The Knowledge Base for Fisheries Management

Edited by
Lorenzo Motos &
Douglas Clyde Wilson
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THE KNOWLEDGE BASE FOR FISHERIES MANAGEMENT

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EFIMAS—Operational Evaluation Tools for Fisheries Management Options is a research project that consists of scientists from 29 European Research Organizations. The objective of the EFIMAS project is to develop an operational management evaluation framework that allows evaluation of the trade-off between different management objectives when choosing between different management options. This book is the main product of the initial phase of the EFIMAS project during which we examined how others had met similar challenges with respect to developing and communicating a useful knowledge base for fisheries management.

The writing and editing of these contributions were carried out in a very open and cooperative manner. The writing team is almost entirely European but beyond this they are a very diverse group of biologists, ecologists, economists, sociologists, anthropologists and political scientists from across the continent. The contributors are experts in many styles of science. Some are primarily model makers, almost to the point that they take a purely mathematical perspective on the issues. Some are qualitative researchers who focus so much on collecting rich information about actual fisheries management experiences one could almost think of them as very slow and careful journalists. Most contributors would put themselves somewhere between these two groups.

Given the complexity of the subject and the diversity of the scientists involved, we chose to give the authors a very free hand in addressing their subject matter. The only requirement was that they reflect on how knowledge is best acquired and used to facilitate the aspects of fisheries management they were addressing. We feel that the resulting chapters paint a rich picture of the many issues involved in creating, collating and communicating the knowledge needed for management. To help the reader navigate through this complexity, a guide through the chapters is provided in the last section of the introductory chapter.

We worked as a team. Chapters were exchanged and reviewed among the authors as well as the editors. An intensive but rewarding meeting was held half way through the process to try to smooth out overlaps and reorganize the flow of the book. The overall experience was an exciting taste of how a truly multi-disciplinary effort can make a, we believe, strong contribution to the reform of fisheries management.

Pasaia, Spain, and Hirtshals, Denmark
Lorenzo Motos and Douglas Clyde Wilson
February 2006
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The EFIMAS coordination team, led by Dr. Rasmus Nielsen, played a central role in making sure that this book actually saw print.

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Chapter 1

Introduction: The Knowledge Base as Process

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This is a book about the knowledge base for fisheries written by a multi-disciplinary team of social, economic and natural scientists for the use of the entire fisheries management community. By “knowledge base” we mean the whole gamut of information needed to make fisheries management decisions. The knowledge base includes biological information about fish stocks, economic information about fisheries, and social information about the requirements of effective fisheries governance. Beyond this already broad list, taking the ecosystem approach to fisheries management requires us to include ecological information of many kinds while the precautionary approach requires us to treat all of this information as being to some degree uncertain. Given this complexity and driven by their own background and experience, the authors take many perspectives on the knowledge base, and we have intentionally avoided heroic efforts to maintain consistency in these perspectives. Our only requirement was that they reflect on the role and use of knowledge in respect to the various issues and approaches they were asked to discuss.

The book had its genesis in a large cooperative research project funded by the European Commission: EFIMAS—Operational Evaluation Tools for Fisheries Management Options. The main purpose of the project is to develop tools, mainly bioeconomic models and associated software, to evaluate alternative fisheries management options. In the context of the Common Fisheries Policy, this objective arises from the current lack of a common evaluation framework tool to structure and communicate existing knowledge in terms of data and knowledge about processes. This tool would enable exploration of options within specific management procedures with an evaluation of both trade-offs between various objectives and the robustness of the options (to assumptions, model and data error) and risks involved. The common framework integrates biological, ecological, economical, and sociological aspects, allowing for the synthesis of complex knowledge in order to help support decision-making processes.

As part of initiating this effort the team, representing 29 European research organizations, decided to review the use of such tools in management (Section 3 of this
book) as part of a broader review of knowledge and fisheries management. This review also included how knowledge in various forms is being used in different approaches to fisheries management (Section 2) and the knowledge-base concerns raised by a set of fisheries management challenges currently being faced in Europe (Section 3). The output of this effort hopefully would be the identification of the context in which fisheries/stocks evaluation tools for production of advice to management bodies are to be used, which problems they are to assist in solving, as well as of the knowledge which the tools should communicate.

By way of introduction the present chapter locates the question of the knowledge base within the recent historical development of ideas about natural resource management. The central argument of the chapter is that the role that the knowledge base plays has undergone a profound shift because our overall understanding of management has itself shifted. One important idea that ties fisheries to the broader field of natural resource and environmental management, and indeed to similar problems in many fields, is the “commons” and the “tragedy” that it is argued to produce. A generation ago a new appreciation of resource degradation and the “Tragedy of the Commons” led to a focus on the design of resource management institutions. This attention to design took a somewhat static form in which the knowledge base for decision-making was treated mainly as an outside input to the management system. During the 1990s the overall discussion began to shift way from institutional design to institutional processes. This shift brought the question of the knowledge base from the margins to the centre of the discussion because what participants in management processes know and how they know it is the primary content of their discussions. When they can make a decision on what is actually going about in the marine ecosystem and the fishing society, the question of what to do about it becomes a great deal easier.

Hence, this book is a timely one, as is the EFIMAS project as a whole. We already have a good idea about what tools are available for carrying out natural resource management. The question is how to combine them into adaptive fisheries management strategies that respond to multifaceted and ever changing ecological, economic, political and social realities. Responses to the current fisheries management challenge, indeed crisis, require new ways of thinking, more comprehensive and flexible scientific models, and innovative ideas about how those models can be used in the development of management strategies.

1.1 THE COMMONS AND WHY WE NEED MANAGEMENT INSTITUTIONS

Fisheries management is just one of a whole group of activities by which people have tried to address the problem of the commons. The problem itself has been long recognized:

For that which is common to the greatest number has the least care bestowed upon it. Everyone thinks chiefly of his own, hardly at all of the common interest; and only when he is himself concerned as an individual. For besides other considerations, everybody is more inclined to neglect the duty which he expects another to fulfil; as in families many attendants
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are often less useful than a few. Each citizen will have a thousand sons who will not be his sons individually but anybody will be equally the son of anybody, and will therefore be neglected by all alike. (From Aristotle’s “Politics”, Written c.a. 350 BC)

A commons, as the name implies, is anything owned by a group. There are many different kinds of commons. Freely available public goods of many types, including music, the internet, scientific knowledge, library books, legal protection, highways, shared housing, public spaces, and urban gardens to name just a few have been analysed as commons. Many natural resources, which by their nature are not owned by individuals, but are shared by a community or group of users, are also commons [1–4]. Our concern in this paper are these natural resource commons, especially fisheries, but it is helpful to understand that many of the approaches we use to understand the fisheries management problem have strong parallels in many other aspects of society.

Natural resources not held by individual private ownership are often analysed as commons, although completely open access resources over which no group or government lays claim are technically not commons. Examples of such resources are wild fish stocks, forests, irrigation waters and pasture lands. Resources, which are diffuse and give unpredictable yields that are low in unit value, are more likely to be commons. They are often in areas that are difficult to divide and/or defend, or are seasonally inundated by environmental changes such as arid areas subject to drought or wet lands subject to flooding [5]. Commons tend to be found where the costs of exclusion are high in relation to the unit value of the resource itself, where this ratio is low then one is much more likely to find a private property regime [6]. Fisheries tend to stand out in that the driving force behind their being maintained as commons is more often the relatively high costs of excluding other users rather than the relative low value of the resource itself.

Whenever a common natural resource is subtractable, meaning that its use by one user implies reduced opportunities for its use by others, some form of management is required. Private property does not require management in the same sense because the owner is responsible for the care of the property, and if he or she fails in this care then no one else is the loser. That a lack of ownership means a lack of stewardship has been an oft-heard policy argument at least since the eighteenth century enclosure and freehold movements in agriculture. Different answers to this lack of stewardship are presented. In recent years a number of investigations and empirical research have deepened the understanding of natural resource management, especially in terms of defining the constituents of particular management systems. Simplistic approaches [7] have defined all commons to be open access regimes while broader scholarship has pointed out that different resources fall under many different kinds of property regimes [3, 6]. It is a complex problem where various solutions have been offered.

1.2 DEFINING THE PROBLEM: THE TRAGEDY OF THE COMMONS

The term “Tragedy of the Commons” (TofC) was introduced to the then emerging environmental and resource management community by an article by Garrett Hardin [7]. Beyond coining this powerful turn of phrase, this was an otherwise mediocre paper that
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foocussed mainly on human population growth, while ignoring almost all of the research on human fertility that had been done up until 1968. The paper has been criticized on many different points, most of them stemming from its reliance on an abstract theory that while sounding very good and logical missed a great deal of the related empirical evidence. The importance of the introduction to environmental discourse of the dramatic phrase “Tragedy of the Commons” cannot be underestimated, however, and the paper serves as an important element in discussions of the commons. It has had a huge impact on theory, policy-making and on defining what issues are at stake when it comes to natural resource management. It is therefore worthwhile to have a look at Hardin’s argument:

Adding together the component partial utilities, the rational herdsman concludes that the only sensible course to pursue is to add another animal to his herd. And another... But this is the conclusion reached by each and every rational herdsman sharing a commons. Therein is the tragedy. Each man is locked into a system that compels him to increase his herd without limit – in a world that is limited. Ruin is the destination toward which all men rush, each pursuing his own best interest in a society that believes in the freedom of the commons. Freedom in a commons brings ruin to all. [7; p 4]

Two basic premises are presented here. First, that the world is limited. This is a trivial statement today, but its relevance stems from the fact that up until the 1960s, absolute limits on natural resources, and therefore on the possibility of overexploitation, had only played a peripheral role in economics and in development theory. And, second, that the commons, in this case an open pasture, are always resources with open access. This, combined with the actions of a rational individual seeking to maximize his pecuniary self-interest, leads to the tragedy due to overexploitation of the commons. The tragedy occurs because the marginal cost of overexploitation is shared collectively while the marginal benefit of, in his example, adding an extra animal is individual. The open access of natural resources are caused by the lack of legally defined property rights to the resource, which leads Hardin to suggest that the commons has to be either controlled by government or transferred into private property.

That there are limits to growth is not a new idea. Resource scarcity was put to the fore in British Political Economy of the nineteenth century when R. Malthus’ essay On population was first published in 1798. His basic argument was that due to human nature, population growth if unchecked would increase at a geometrical rate, while subsistence could only increase at an arithmetical rate. This would lead to overpopulation, misery and hunger. For Malthus the solution was to dissolve the British Poor Laws to make sure that nobody would procreate beyond self-sustainable levels—in other words, there would be less poor people, because their children would not be able to survive if the State stopped subsidizing them. The scarcity Malthus warned against was relative, i.e., a function of the relationship between population and food production, and as it turned out, had a technical solution, most importantly being more efficient ways of food production. Also more thorough empirical investigations of actual population growth proved Malthus wrong, but the notion of a fundamental scarcity of resources as an intricate part of the human condition as such had entered the political and economic thinking [8].

When S. Jevons published The Coal Question in 1865 he transformed scarcity into a notion of absolute scarcity, when he called attention to the gradual exhaustion of Britain’s
coal supplies, by applying the same methods of investigation as Malthus had used. Despite his engagement with the question of absolute scarcity, Jevons did not relate the issues presented in *The Coal Question* with his marginal utility theory. Instead he, along with Marshall, Menger and Walras, strived to formulate a precise mathematical methodology that became the foundation of neoclassical economics. Even though this methodology greatly increased the exactness of economics it also represented the issues of scarcity and resource exhaustion in strictly economic terms, i.e., as externalities or questions of resource allocation through price mechanisms. As a consequence, it also defined the solutions in strictly economic terms such as defining and enforcing property rights, internalising externalities and so on. In this perspective property rights, and the institutions upholding them, plays the central role in relation to natural resource management.

Clearly the formulation of the problem as the TofC lends itself well to economic reasoning and modern economics has adapted and developed this model resulting in a number of theoretical insights and institutional responses. In an open access situation, with no property rights or any control of access, the dilemma between individual gain and the common cost is likely to lead to free-riding and negative externalities. Free-riding would occur because no incentives in the form of coercion or other property rights enforcement would stop the individual from externalizing the costs of overexploitation to the community. Subtractability and high costs of exclusion lead to a number of different problems or negative externalities, which occur whenever the marginal cost of an individual’s action is less than the marginal cost that society incurs for the individual taking that action. Following Ostrom et al. [9] negative externalities can be grouped as appropriation and provision problems. Appropriation problems appear as issues of excessive appropriation, congestion, competing technologies arising when beneficiaries are presented with a dilemma of how to exclude others and allocate the resource. Provision problems results when beneficiaries are faced with the problem of maintaining or developing a resource and preventing it from depletion. Provision problems can emerge both from the demand and supply side.

Fishing is an excellent example of a resource in which the TofC model is applicable. Any commercial fishery may exhibit several types of the above-mentioned externalities [9–12]. Nevertheless, the fundamental one is the appropriation externality derived from the resource base itself. The stock is a limited and free factor in each firm’s production function. Thus, each firm, by fishing, reduces the harvesting possibilities of others. Demand side provision problems resulting from the excess demand of harvesters fishing at a rate greater than the rate of reproduction are also a common matter of commercial fisheries. Extracting at such rates may have a net negative effect on future populations and thereby reduce the available catch in future years. Externalities generate the so-called fisheries problem, which manifests itself as excessive fishing capital and fishing effort, reduced fish stocks, dissipation of economic rents and social welfare losses. As everybody can take freely from valuable stocks (leaving less to others) first come first served strategies will be generally adopted, resulting in capital stuffing and efficiency losses. With a complete system of property rights guaranteeing exclusion in place taking would be permissible only for those owning the right to do so; non-excludability would then be overcome. It follows that the fisheries problem would disappear if the appropriate property rights could be defined, imposed and enforced. But things are not as easy as that.
There are substantial biological, economical, technical, social and political difficulties to defining, imposing and enforcing sufficiently good private or communal (or group) property rights above the fishing resources.

Property rights take many complex forms. Ciriacy-Wantrup and Bishop [13] pointed out the need to distinguish "common property" from "everybody's property", the latter being no property rights at all. Common property encompasses a wide variety of property regimes different from open access (cf. [6, 3]). In relation to natural resource management, a variety of property rights regimes are at play in different situations, and in relation to different commons. Empirical investigations of existing community arrangements show that community-based management systems does not necessarily lead to resource degradation and over-exploitation [4]. Bromley [6] argues that property, when it comes to common resources, is not a universal and immutable classification: "It is essential that property is not an object such as land, but rather is a right to a benefit stream that is only as secure as the duty of all others to respect the conditions that protect that stream" [6]. This means that "property is a triadic social relation involving benefit streams, rights holders, and duty bearers" (op.cit.). Property is thus seen as a social instrument and the particular property regime becomes dependent on the nature of the particular resource and the stakeholders involved in it. Bromley suggest four different types of property rights regimes, [6]:

**State Property**: Individuals have *duty* to observe use/access rules determined by controlling/managing agency. Agencies have *right* to determine use/access rules.

**Private Property**: Individuals have *right* to undertake socially acceptable uses, and have *duty* to refrain from socially unacceptable uses. Others (called “non-owners”) have *duty* to refrain from preventing socially acceptable uses, and have a *right* to expect that only socially acceptable uses will occur.

**Common Property**: The management group (the “owners”) has *right* to exclude non-members, and non-members have *duty* to abide by exclusion. Individual members of the management group (the “co-owners”) have both *rights* and *duties* with respect to use rates and maintenance of the thing owned.

**Non-property**: No defined group of users or “owners” and benefit stream is available to anyone. Individuals have both *privilege* and *no right* with respect to use rates and maintenance of the asset. The asset is an “open access resource”.

Each of these types is a fairly simple categorical ideal holding many different practical forms by which access to benefit streams can be defined and controlled. There is no evidence that any of these regimes is best suited for all types of commons [14] but a continually growing body of research has emerged about how this set of social relationships plays itself out on different types of commons in different cultures.

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1 The interested reader is referred to the International Association for the Study of Common Property <http://www.iascp.org>
1.2.1 Implications for the knowledge base

This emphasis on the TofC and property rights had important implications as far the knowledge base for management is concerned. With the rise of the TofC as the new definition of the problem, resource and environmental management for the first time was seen to involve a major role for the social sciences in management, a field which hitherto had been seen almost exclusively as a technical, natural scientific process. Now the problem was being defined in terms of property rights and property rights are a cultural construct. Historical and social, as well as biological, knowledge was now seen as important to the knowledge base needed for management. These aspects of management knowledge had their own research problematic: institutional design.

1.3 THE ELEMENTS OF INSTITUTIONAL DESIGN

The research on institutional design was about developing the tools for natural resource management. Social scientists use the term institutions in a slightly different way than it is used in common parlance. We use the term to point to shared meanings that pattern human behaviour in a way that persists over time. This can mean big institutions like the family or an art gallery, but smaller behaviour patterns can also be analysed as institutions; including natural resource management tools that are meant to pattern the ways people behave in respect to their environment. Institutions consist of cognitive, normative, and regulative structures and activities that provide stability and meaning to social behaviour [15]. Different perspectives have different definitions of institutions and the role they play in a management situation, but the key point here is that institutions affect social behaviour in different ways. Some of these reflect the neoclassical image of an atomized, rational individual, indeed a particular sub-set of economic institutions relies on making this kind of behaviour normative. In other words, business people may work hard because they love their families, but if they fail to make business decisions in an economically rational way they will fail to achieve this altruistic goal. Different institutions demand different kinds of behaviour.

Environmental management institutions can be characterised along three dimensions:

(a) The degree to which they incorporate hierarchical governance mechanisms: Most natural resource management regimes in developed countries that deal with commonly owned resources like fisheries are most fundamentally a form of hierarchical governance in which a central agency representing a government makes management decisions, which have the force of law and which are enforced by government agencies. Other management institutions are usually rooted in, and take place within, the hierarchical system. The reasons for this are threefold. First, and most fundamentally, when natural resources are commons they very often belong to all citizens, and it is the responsibility of the government to manage those regimes on their behalf. Second, hierarchical governance systems are the most effective basic approach to the management of resources that cover a large geographical scale because they produce relatively predictable outcomes across wide areas. However, they pay an important price for this in both local legitimacy and support
and having to make decisions based on much poorer information than is available on the smaller scales [16]. Finally, hierarchical governance systems are able to respond and deal with problems where negotiated outcomes are difficult to achieve. Where management faces problems with multiple jurisdictions and competition over resource allocation, there are simply decisions that are best made by central authorities.

(b) The degree to which they incorporate market-oriented governance mechanisms: The great strength of market-oriented governance mechanisms is that they are able to produce a spontaneous order from decentralized decisions governed by the laws of supply and demand. For this they are dependent on well-defined property rights. Markets allow the knowledge of the whole society about certain simple questions to be mobilized—hundreds of different kinds of screws exist, and the market is able to accurately determine how many of each kind should be made. The same power applies to fishing decisions so that markets are able to determine how much of what kind of fishing should be done to maximise the material benefit of the whole society. Markets work through competition that is expressed through the allocation of resources at the margin: whoever makes the best decision about the investment of the last fishing boat bought or last hour of fishing effort will out-compete his rivals. The problems with markets as governance mechanisms appear when the basic assumptions of the market model do not apply for political, social or ecological reasons. When margins are large or irreversible the market is much less effective at allocating resources. Certain types of commodities such as labour, and sometimes land and natural resources, defy the assumptions of the market model because they cannot become valued at zero if necessary to clear the market [17]. Markets depend on actors seeking to maximise their profits while natural resources have natural limits. Markets also undermine themselves in the long run because of the tendency for competition to be self-eliminating as losers go out of business and the corrosiveness of economic self-interest on the moral context on which the market depends [18]. And, centrally important to the current discussion, markets fail when property rights are weak or unclear.

(c) The degree to which they incorporate civil-society governance mechanisms: In their discussion of fisheries co-management Jentoft and McCay [19] take from Walzer [20] the following definition of civil society: ‘the space of uncoerced human association and also the set of relational networks formed for the sake of family, faith, interest, and ideology—that fill their space’. The central importance of the civil society governance derives from the advantages of communications and information sharing made possible by a richer set of relationships than those presupposed by the market or hierarchical governance approaches alone. Both bureaucracies and markets depend on semi-automated communications that simplify interactions, with the former depending on commands backed up by sanctions and the latter on ‘take it or leave it’ exchanges. All of these involve some degree of coercion, which makes it possible for the institutions to function predictably over large scales. For this reason they communicate poorly the kind of complex information that requires background information and question-and-answer interactions that seek to convince people that something is true. Hence they must be supplemented by civil-society mechanisms—such as networks, small scale organizations, and interactive fora—which make it possible for participants in management to exchange rich information [16]. Such
information is critical to the enforcement of management measures, and it is also crucial to the development of an accurate knowledge base. Science is likely more dependent on the communication of rich and complex information than any other activity. The drawback of civil-society governance is that it works poorly across large scales. Nested systems and other forms of representation are the institutions that structure civil-society governance over large scales, and while this is possible to some degree representation has to be supported by top-down rules of selection which reintroduce the drawbacks of hierarchical governance.

Much of the debate around the design of management institutions over the years reflected the emphasis people wanted to put on these different kinds of mechanisms. All of them play a critical role. Because hierarchical governance is both legally foundational and often quite clumsy in practice the debate has tended to both assume its importance while making it the focus of the critique. Critiques from both the market side (property-rights approaches) and the civil society side (community-based approaches) have tended to focus on hierarchical (top down management) as their primary target. We will continue with this assumption about the foundational role of hierarchical governance and in the next sections focus on the development of the other two approaches to institutional design.

1.4 INSTITUTIONAL DESIGN I: PROPERTY RIGHTS AS THE BASIS OF GOOD MANAGEMENT REGIMES

Once the problem had been constructed as “the ToC” the earliest responses were developed based on the premise that the absence of or badly defined property rights is the key to a good understanding of the fisheries problem. Property rights may be defined as the de jure and the de facto rights of individuals or groups of individuals to a flow of benefits from assets, with at least a partial right to exclude others [21]. Many other definitions of property rights exist including rights of individuals to use the resources [6, 22].

Economists and others have done extensive research into the properties and management potential of property rights institutions. Beyond the complexities introduced by the different types of property rights mentioned in the last section, a property right itself is something with important internal dimensions; it consists of a collection of several characteristics such as security, exclusivity, permanence and transferability, which, exerted in different degrees, determine the completeness of the property rights. Transferability refers to the ability to transfer the property right to someone else. For any scarce resource, this characteristic is economically important because it facilitates the optimal allocation of the resource in the hands of the more competing users. An important feature of transferability is divisibility, i.e., the ability to subdivide the property right into smaller parts for the purpose of its transfer.

Property rights do not have to be considered as exclusive use rights. Exclusive use rights do not necessarily confer ownership of the resource itself; rights are limited to the resource flow. Consequently, rights may have more uncertainties than conventional economic goods, due to natural events and also to changing regulations and economic conditions. Nevertheless, exclusive use rights may fulfil the two basic conditions needed
for the existence of some sort of rights over the fishing resources: the exclusion of those that have no rights and the protection of the rights by authority. The theory of property rights argues that the more rights the resource-user holds, the greater the incentives to internalise the social shadow value of the fishing resources in the decision-making process. In other words, owners will be more encouraged to achieve efficient and conservationist solutions than proprietors, claimants and authorised users. Owners hold all the operational and collective choice rights. Proprietors only lack alienation right, while claimants possess operational rights plus management right. Authorised users only have access and withdrawal rights.

Property rights take different forms in different cultures. There are abundant examples of traditional fisheries all around the world where fishermen have been able to face (with more or less success) the fisheries problem through community rights and excluding non-members through common property systems emanating from the local community [9]. Some property-rights regimes shift over time in response to changes in the resource. Movements to and from common property systems can also be identified and no unidirectional tendency seems to exist for commons to become private. Private property has been known to become common property, and property regimes may shift back and forth in long, often seasonal cycles [5]. Bauer [23] describes a land tenure system in Ethiopia that moves cyclically between a common and private property system following decisions made at community meetings. When the community decides that it needs more people it declares the agricultural land to be common, and when it decides that there are too many people it divides the land up as private property. However, several mechanisms for the breakdown of these common property systems have been identified. Johannes [24] attributes the breakdown of traditional management of marine common property in Polynesia to commodification of the economy, colonial government policies, which did not recognize the importance of restricting free access to fishery areas, and the breakdown of the traditional authority on which the traditional practices had depended. McCay [5] offers a long history of the breakdown of common property system in the West as a process of powerful economic forces disenfranchising the more vulnerable portions of the population.

Given this kind of complexity it is helpful to think of property rights as a bundle of attributes. In fact, right-based management systems exist as a continuum in terms of their most important characteristics, i.e., transferability, exclusivity, security, durability. Regarding the nature of the underlying property, there are three kinds of rights units: area, inputs and outputs. Rights holders can be individuals (persons, vessels owners, firms, etc.) or groups (cooperatives, producer organisations, fishermen guilds, communities, etc.). The time horizon or duration of the rights (i.e., a season year, multi-year and permanent) and all the issues concerning to transferability (i.e., initial allocation, eligibility and limits to transferability) complete the core of the broad spectrum of right-based systems.

Just as there is no ironclad rule that a rights-based management system will manage a particular resource better than one of the other types of management, there are no ironclad rules about which type of denomination (area, input or output) or eligible right holder (individual, group) will work best when developing a rights-based system. Examples of
Rights-based management institutions in fisheries include territorial use rights, fishing input rights and fishing output rights.

**Area or territorial use rights** in fisheries (TURFs) conveys the right to fish within a specific area. Such rights could be limited to the use of particular gear types or species. Most of traditional fisheries management regimes are based in territorial group rights. For example, the mentioned instrument is the main supporting of the Japanese inshore fisheries management regime and is increasingly claimed by small-scale fishermen’s unions and communities in many African and Pacific countries. An interesting example of territorial individual rights can be found in the mussels fisheries along the Rías Baixas in Galicia, Spain. Area rights may work very well in sedentary fisheries, but will have many drawbacks in migratory fishing stocks.

**Fishing inputs rights** granted the holder the right to use certain inputs or fishing gears, frequently in selected areas and/or fisheries and at specified times. The right is based on physical harvesting capacity and is measured in terms of number of licensed boats and/or attributes of boats such as units of traps, boat days, etc. Since stocks may vary over time due to environmental fluctuations, it is also convenient to control the use of any licensed fleet from time to time. Particular well-known examples are the Faeroe Islands system of individual tradable fishing days, the Western Australian lobster fishery (where the unit of ownership is the individual lobster trap) and also the Australian northern prawn fishery. Group input rights are found in Japan, Norway, Gambia, India, Senegal and Sri Lanka. Limits on the number of vessels and fishing-trips and hours fished are common management measures for Japanese cooperatives. In the Norwegian Lofoten fishery, fisheries cooperatives undertake various regulatory functions, primarily based on input-limitations as well as technical management measures such as closed seasons and areas.

Rights can be denominated in terms of outputs. In this case, right holders are allowed to harvest a specific amount of fish each year or season. The most extended ones within the broad range of output based systems are the well-known individual transferable quotas (ITQs) being applied since the middle 1980s in New Zealand, Australia, Canada, Iceland, Chile, USA and the Netherlands. Examples of output-based group rights are found in the Netherlands, New Zealand, Philippines, Senegal, UK and also in USA. In the case of the UK, producer organizations have been given the right to distribute quotas among their members. In the Netherlands quotas have been allocated through the ‘grandfathering’ principle, through which the allocation of IQs is done by taking in account historical catches of the fishing enterprises, and represent a percentage of the national quota. Dutch fishermen pool their ITQs in co-management like groups, in order to monitor the uptake of the quota and to trade and rent out quota smoothly. In New Zealand Maori have been assigned both territorial and quota based fishing rights. In the Atlantic coast of USA, community development quotas have been allocated as part of an ITQ regime.

In choosing between input- and output-based rights, the second has two important advantages. Quota regimes are TAC based; therefore they have the potential to directly achieve biological objectives. The control of fleet capacity and effort only offers an indirect control of the catches. In addition, quota programs provide incentives to choose the cost minimising input combination, while input controls can provide incentives to use
non-restricted inputs if these will increase harvest, resulting in higher costs. Furthermore, since technological progress may increase the capacity of the fleet beyond optimal levels in a relatively short time, it stands that paradoxically the more successfully the fishing capacity is controlled the more the industry may be denied the gains of technological progress. However, these advantages of the quota systems are balanced with increasing incentives to discard fish, difficulties to set credible TACs (specially in presence of by-catches), and the complexity of monitoring the individual harvests of many participants landing fish over widely dispersed areas.

1.4.1 Rights-based approaches and the knowledge base

Rights-based approaches to fisheries management have some important implications for the knowledge base. Vesting property rights in a resource in principle also invests the responsibility for its ongoing management, including the responsibility for developing the knowledge base. In ITQ systems in New Zealand and the United States the quota holders have direct financial responsibility for fisheries research. This is another example of internalizing an externality, in this case the cost of the knowledge is internalized as part of the costs of doing business rather than externalized to the public purse.

The reliance of rights-based approaches on resource units to which rights attach will have implications for the content of knowledge base. In the case of ITQs the system is forced to maintain a quota system. ITQs themselves do not eliminate the problems with bycatch, under-reporting catches, high-grading and other forms of discarding that so often undermine the development of good data under TAC systems. Indeed, an ITQ system may lock in the TAC approach even when it is formally granted as a temporary right because the transferability would make any change in the basic system very complicated.

The need for units to denominate rights will tend to focus research on those aspects of the marine ecosystem that are most closely associated with the right. In the case of ITQ systems paying for their own research the quota holders are mainly interested in paying for research to determine the TAC of the species they own access to, not other aspects that might be important for an ecosystem-based approach to fisheries management. The knowledge base implications of rights-based systems should be considered in the institutional design anticipating and avoiding these tendencies to narrow the scientific problem.

1.5 INSTITUTIONAL DESIGN II: COMMUNITY APPROACHES

Although property rights are generally recognized as the central element in natural resource management institutions several other perspectives have been introduced mainly in respect to the role of the civil society in environmental governance. Because over-exploitation and resource degradation also happens in other forms of property regimes than open access the provision of property rights is not enough [3]. Instead of focussing exclusively on property rights regimes, attention has turned toward other institutions and management systems surrounding the commons. Why do certain institutions manage a common pool resource well while others do not? The main focus has been on mobilizing
the dynamics of “community” for fisheries management. It is a common mistake to think that community perspectives reflect a belief in the power of altruism, this is not the case and, indeed, aid in the enforcement of measures is often a primary motivation for community involvement. All institutions must have recourse to various forms of sanctions because on a commons users will not adhere to a conservation norm even if they want to if they have no reason to think that other users will do so.

Community approaches are often also property-rights approaches when they are based on the groups having a kind of shared property right to the resource. Even a commons management system based on exclusive use rights to the resource implies a division of both rights and tasks between governing institutions and the users. When comparing individual and community (or any group) rights, an important advantage of group fisheries management systems lies in the potential of lower transaction costs related to management, i.e., savings in information, monitoring and enforcement costs through the use of information held privately by fishermen and the use of social capital embedded in local and professional organizations and institutions. Furthermore, the best knowledge of the individual and collective preferences facilitates achieving mutually satisfactory management objectives. Consequently, there is greater likelihood that rights holders respect and comply with management rules that were designed and agreed upon by them. The potential shortcomings of group right systems frequently derive from insufficiently specified, exclusive and protected group rights, which are not recognized in formal law and are inadequately protected from external threats that erode long-term stewardship and legitimacy. Sometimes group rights simply may fail as a result of weakness in internal governance. When management rules are not able to accommodate technological progress, natural population growth, or market constraints the pressure of the growing group can drive to open access allocations, especially where there is a lack of alternative livelihoods in other sectors of the local economy.

A common denominator for the community-based approaches is the notion of embeddedness. The basic point, as Polanyi put it, is that “man’s economy, as a rule, is enmeshed in his social relationship” [17]. Or as McCay and Jentoft put it:

Contrary to the neo-classical and ‘new institutionalist’ economic perspectives which see rational behaviour as motivated by desire to maximize individual gains, the embeddedness perspective would regard rationality itself as ‘anchored’ within the social context”. The embeddedness perspective “... leaves open the question of what the cause of any particular tragedy is, but which by emphasizing embeddedness opens up the possibility of ‘community failure’ as an important cause. The question is shifted from the existence of one or another form of property rights to why some communities succeed in preventing or ameliorating problems in the use and management of common resources and others do not. [2]

User groups and stakeholder involvement play a central part in the design of institutions managing a commons. Extensive empirical investigations have been carried out in existing community arrangements within natural resource management areas of irrigation, fisheries and grazing [6, 4]. From these investigations emerged a variety of approaches and terms that sought to describe well-functioning management systems on different commons.

Ostrom [4] focuses on self-organising institutions and self-governance in commons situations in order to discuss how institutions can be designed to support collective action.
Schlager and Ostrom [26] make a very useful distinction between rights at the operational level (i.e., the rights of access and withdrawal) and rights at the collective-choice level (i.e., the rights to make management decisions, to exclude and to buy or sell). Operational level rights holders merely adjust their strategies to the rules that define their rights; they cannot modify them. The rules governing the operational level are decided and changed at the collective-choice level, by collective-choice actions undertaken within a set of collective-choice rules that specify who may participate in changing operational rules and the level of agreement required for their change. The authority to devise future operational level rights is what makes collective-choice rights so powerful.

Based on extensive comparative research, Ostrom offers eight design principles that account for much of the success and robustness of long-enduring commons institutions. The principles are [4]:

1. Clearly defined boundaries: Individuals or households who have rights to withdraw resource units from the commons must be clearly defined, as must the boundaries of the commons itself.
2. Congruence between appropriation and provision rules and local conditions: Appropriation rules restricting time, place, technology, and/or quantity of resource units are related to local conditions and to provision rules requiring labour, material, and/or money.
3. Collective-choice arrangements: Most individuals affected by the operational rules can participate in modifying the operational rules.
4. Monitoring: Monitors who actively audit commons conditions and behaviour are accountable to the appropriators or are the appropriators.
5. Graduated sanctions: Appropriators who violate multifaceted rules are likely to be assessed graduated sanctions [...] by other appropriators, by officials accountable to these appropriators, or by both.
6. Conflict-resolution mechanisms: Appropriators and their officials have rapid access to low-cost local arenas to resolve conflicts among appropriators or between appropriators and officials.
7. Minimal recognition of rights to organize: The rights of appropriators to devise their own institutions are not challenged by external governmental authorities.
8. Nested systems: For commons that are part of larger systems, appropriation, provision, monitoring, enforcement, conflict resolution, and governance activities are organized in multiple layers of nested enterprises.

These principles symbolize and to a large extent summarize the results of the community-based approach to institutional design. They have also given birth to a research agenda. In these principles there are a number of variables that are dependent on the particular commons and the property right regime. Depending on the size of the commons (local, regional, global), institutions are required at various decision levels, and the possible linkages between them, both vertically and horizontally, needs to be investigated [14]. Observations about the importance of scale and institutional linkages, and the multifaceted related variables that bear directly on how stakeholders interact, has led to the change in research emphasis from design to process.
1.5.1 Community-based approaches and the knowledge base

These community-based institutions have direct implications for the knowledge base. One part of their importance is access to local people’s experience-based knowledge (EBK) about the resource. McCay [27] points out that indigenous common property management systems often “encode and are based upon intricate, detailed, accumulated knowledge and wisdom about marine ecology”. Freeman [28] argues compelling epistemological reasons for believing that local management systems may have advantages over the standard systems. He writes that “a common feature of many of the traditional management systems... is that the harvesters have access to a lengthy time-series of data by species, season and locality, in addition to extensive information that tends to emphasize relationships between species and various environmental parameters”. The local approaches rely upon data and techniques of analysis that local resource users can control and utilize in real time, so as to take locally sanctioned corrective actions with a minimum of delay. This kind of reasoning is especially important as we move toward an ecosystem approach to fisheries management.

But the question of civil-society and knowledge base goes well beyond experience-based knowledge. The participation by fishers in collaborative research and other forms of interaction with scientists is critical for the development of a knowledge base that will be effective for management [29]. Civil-society aspects of formal scientific institutions can also be very important [30]. The bottom line here is that forming an effective and comprehensive knowledge base is heavily dependent on the civil-society aspects of management institutions functioning well.

1.6 INSTITUTIONAL PROCESSES: A NEW ROLE FOR SCIENCE

During the 1990s, social scientists studying natural resource management have begun to move away from institutional design as the central research agenda, mainly because the institutional design approach seemed to have made its most important contributions. In fisheries, the 1980s saw the development and initial implementation of the individual transferable quota, the design innovation that best expresses the insights from the strong property rights approach of fisheries economics. Publications by Pinkerton [31] and Jentoft [32] in particular established fisheries co-management as a useful general approach. Both ITQs and co-management are now widely accepted. In the broader resource management field, Ostrom’s [4] design principles have taken on a sort of iconic status among social scientists. In the 15 years since they were published they have been the basis of many research projects and reports focussing on such questions as the conditions under which particular principles take on greater importance or how they can be best implemented.

Recent criticism of the products of the design approach has not been so much about the contents of particular institutional innovations or design principles—there have been no major arguments about how X principle should be added to Ostrom’s list or that Y principle should be taken away—as about the idea of design principles itself. Steins et al. [33], for example, argue that the design principles approach contains two serious flaws. The first is simply that the principles can be interpreted as a prescriptive blueprint
applicable to any situation, a criticism more of their misuse rather than of the principles themselves. This kind of misuse of ideas for institutional design is not uncommon. Several institutional innovations have garnered a following of people who promote them as the way to “solve” the management problem. In respect to fisheries, Degnbol et al. [34] point to marine protected areas from conservation biology, individual transferable quotas from fisheries economics and participatory co-management from sociology and anthropology as examples of tools that have become common prescriptions and among some people there are even movements to promote them.

The second flaw Steins et al. [33] point to is that a focus on design turns our attention away from the external factors that affect how management institutions will function, in spite of the fact that the external environment even “supplies the stakeholders” (pp. 2) involved in management. This argument should not be overdrawn. Ostrom’s principles, for example, emerged out of an empirical investigation of a large number of resource management institutions, all of which were obviously dealing with external environments and the principles should be understood as lessons about what kinds of institutions do that well. Nevertheless, the general insight Steins et al. [33] are making is an important one. One of the basic lessons that have emerged from social science studies of natural resource management it is a political process that is often inaccurately constructed as a technical problem [35]. A design principles approach, when used prescriptively and in a way that reduces attention to external political dynamics, merely reinforces this inaccurate perception.

As a result of both these achievements and these criticisms, during the 1990s social scientists looking at natural resource management institutions have swung heavily in the direction of examining institutional processes rather than design. This is very much in keeping with developments in the broader resource management field such as the precautionary principle and ecosystem-based approaches. At the centre of this process paradigm is the question of institutional adaptation, i.e., how do real institutions formed through political processes manage to change and adapt in response to their environment, especially their ecological and economic environments. Carlsson and Berkes [36] argue in respect to fisheries co-management the emerging focus should be on networks, social power, institutional learning and capacity building. Others have pointed at the central questions of conflicts and institutional scale [16]. Theoretical tools being used in this effort include actor network theory [33], complex systems theory [37] and communicative systems theory [30, 16].

How do we deal with the fact that institutions are neither static entities, nor do they function in static environments? They are much more than something that one designs, they are constantly changing processes that require governance. Now we do no longer think of the politics of management as a problem coming from the outside, but as the process itself.

This change has brought the key questions treated in this book, those dealing with the development and use of the knowledge base, from the periphery to the centre of natural resource social science.

When the question was institutional design the assumption was that scientific knowledge was an input coming from outside the management institution and defining the decisions that the institution had to make. Understood as a unit of institutional design,
the use of the ITQ simply assumes that defining a TAC for the species is scientifically feasible. Co-management systems define a set of responsibilities in which scientists evaluate the resource while making use of the fishers’ knowledge as an input. When we view these things as processes, however, the implications of TAC’s for the development of the future knowledge base becomes a central problem, as to the differences in perceptions of the resource between fishers and scientists in the co-management system.

When the emphasis was on institutional design then it could be assumed that science would be a source of objective knowledge about which all would agree and decisions would be made. When management institutions are seen as an interactive process, however, the central goal of creating the knowledge base is no longer objective knowledge, because from a process viewpoint there is no such thing. From a process viewpoint all knowledge arises from some place and exists in a form that someone has chosen. The central question shifts from objective knowledge to transparent knowledge, because an effective management process requires that participants account to one another about how they know what they say they know. Indeed, this is perhaps the central activity of natural resource management institutions. If a working agreement can be reached about some approximate truth on which to base decisions, actually making the decision is often the lesser challenge.

This de-emphasis on objectivity may seem at first glance to undermine the authority of science. This would be a misinterpretation. On the contrary, what undermines the authority of science in natural resource management is the constant pressure on scientists to produce objectivity out of what they know to be deep uncertainty [30]. We are also not suggesting that objectivity is no longer an important value, (the relationship between the roles of objectivity and transparency are discussed in more detail in Chapter 13) rather that when the focus is on maintaining a process of adaptive management then objectivity must be thought of as a claim rather than a fact. The mainstream idea that scientists can deliver objective advice to a decision-making process from which they are otherwise divorced is a naive one. The emphasis on transparency returns natural resource science to its roots as a set of norms and methods that require transparency so radical that, at their limit, knowledge claims should be so clear that everyone is able to reproduce the circumstances that produced them. Hence, scientists must still take the lead in developing the knowledge base.

Put into practice in fisheries this shift would require changes in emphasis and role more than on the overall scientific activities. Scientists would continue to measure, theorize and model the dynamics of ecosystems, fish populations and economic activities. The difference would be in how these models would be used to focus the management process. They would structure decision steps in stakeholder negotiations over management strategies rather than simply providing fast numbers for negotiations to divide among user groups. In such a use the implications of the various sources of uncertainty around the models would be a central focus of the discussion rather than a peripheral concern. Scientist would facilitate the management process, participating in the discussion and holding people accountable for their claims. This would take place through participation in interactive fora, collaborative research and research into experience-based knowledge as well as at the centre of negotiations over management strategies.
Wilson et al.

The approach to building evaluation tools taken by the EFIMAS project reflects the beginnings of efforts to find ways for scientists to play this kind of role within processes of creating management strategies. Bringing in their expertise from a number of disciplines the scientists who contribute to this book are all coming at this single question from a number of different perspectives: what is the most useful way to provide the needed biological, ecological and economic knowledge to the fisheries management process?

1.7 AN OVERVIEW OF THE BOOK

Because of its origins within an effort to improve European fisheries management “The Knowledge Base for Fisheries Management” focuses on questions of interest to Europe, which does not undermine the interest for people and organisations elsewhere. The book consists of three main sections preceded by this ‘Introduction’.

Section 1 is entitled ‘Global Experiences with Management Systems Relevant to Europe’. It consists of six chapters each of which reviews a different basic approach to fisheries management with special attention to the implications of the approach for the knowledge base. Each chapter consists of an overview of the particular approach to management and a brief, empirical discussion of the strengths and weaknesses of the approach as they have been implemented. The remainder of each chapter describes the implications of the approach for how scientific advice is created, disseminated and used in decision-making.

Section 2 shifts the focus from approaches and techniques to the knowledge base implications of specific management issues currently important in Europe. The first chapter describes the European system in the same way that the generic approaches are described in Section 1. Then a series of issues are addressed from the perspective of the role that scientific knowledge plays.

Section 3 then addresses the narrower question of scenario modelling and the way it can be used as support for fisheries management strategies evaluation. Scenario modelling is becoming increasingly important as a technique for relating aspects of the knowledge base to decision-making in a way that makes assumptions explicit and focuses discussions among stakeholders. This Section 3 specifically presents and discusses Evaluation Frameworks of Fisheries Management Procedures and Management Strategies because of the fundamental importance of this topic with relation to the objectives of the EFIMAS project, i.e., to set up an operational tool for the evaluation of management strategies for European fisheries. The Section includes a first chapter with the methodological review of Management Procedures and the methods used to evaluate them. A second chapter presents an analysis of the implementation of evaluation frameworks already put in practice in the context of the production and use of knowledge, including participation of stakeholders.

In order to provide a guide to the reader, and to highlight the main issues discussed and concluded in the book, in the remainder of this section we briefly summarize the chapters, mainly through paraphrased excerpts from their conclusions.

In Chapter 2, Martin Aranda, Arantza Murillas and Lorenzo Motos begin our discussion of relevant management systems by examining experiences with the management
of shared stocks. They find that cooperative arrangements are the best means to manage shared stocks through time and these systems rely on and are facilitated by shared knowledge. The inclusion of user’s knowledge in this context is emphasized. Of particular interest are biological knowledge related to the distribution of migration patterns and stock structure and economic information such as costs of harvesting and value of the catches. This information must be supplemented by additional social knowledge, including consumer preferences and the socio-economic importance of the shared stocks in different locations/countries and the fishers’ harvesting behaviour. Knowledge of social, technological and other bio-economics factors should also be considered as an important input in the difficult allocation process in cooperative management of shared stocks. Beyond the knowledge needs for this kind of management systems, the importance of an appropriate control and enforcement system is a cornerstone in the way to assure a sustainable exploitation of the resources by the participants.

In Chapter 3, Ikerne del Valle, Ellen Hoefnagel, Kepa Astorkiza and Inma Astorkiza examine developments in rights-based management. While ITQs reduce fishing effort they generally do not reduce it to exactly the optimum level partly due to biological and/or bio-economic uncertainties in setting the total allowable catch (TAC). As in other systems, when fishermen’s knowledge (i.e., perception of the resource) is not incorporated in the decision making process, legitimisation is also difficult in an ITQ system. Cost, scale and chaos problems are identified in the generation, communication and utility of the knowledge. The authors argue that the role of biological science and the research based knowledge has remained dominant, while movement has been seen toward privatizing research and even using information contained in quota market prices to inform the settings of TACs on which to base future quotas. This movement toward the privatisation of knowledge production also contains elements of more collaborative ways of producing knowledge.

In Chapter 4, Ellen Hoefnagel, Amy Burnett and Douglas Clyde Wilson address the knowledge-base issues around government and fisher cooperation through co-management programmes. They examine the problems that arise because of the different forms that fishery knowledge can take. They provide a large number of examples of collaboration efforts that range from efforts where fishers defer almost entirely to scientists in the development of the knowledge base to other examples where they play nearly equal roles.

In Chapter 5, Inma Astorkiza, Kepa Astorkiza, Hans Frost, Erik Lindebo and Ikerne del Valle review the theoretical literature about trade distortions and correction of market failures, with particular attention to the complex role of subsidies and taxes. The questions raised by this literature for the knowledge base are then evaluated within the broader bio-economic context. They evaluate the knowledge base with respect to institutional structure, management procedures, models and data sources as well as scientific advice and its communication. In order to ensure an economically optimal fishery, they think it necessary to have full and detailed information about the cost structure of the industry, fish stocks, catch compositions including discards, and market prices. They advocate the use of models to alleviate the strong demand of data the bio-economic framework demands. Such models need to include biological and economic aspects and reflect that supply is backward-bending as a function of increasing fishing effort or fishing mortality. The authors remark that these characteristics of the fishing industry are different from
almost all other industries and lead in many cases to different conclusions. In addition to bio-economic data, it is also necessary to have data regarding financial transfers, including the public costs of General Services (monitoring, enforcement, research, etc.). However, data collection in this area needs further development and reinforcement. The authors argue that in Europe a general lack of economic information has made it difficult to implement the recommendations stemming from this literature. An ambitious program has been agreed in the European Union regarding compulsory biological and economic data collection at the Member State level. This program aims at harmonizing and expanding the existing data collection that already takes place in many Member States. The use of financial transfers is well documented by the EU’s structural policy, and in this area the EU seems to be at the forefront on a worldwide scale. A final corollary is that any improvement in this sphere is associated with a more holistic use of this information with respect to trade distortions, environmental disruption, and distributional effects including cohesion.

In Chapter 6, Martin Aranda, Arantza Murillas and Lorenzo Motos evaluate the most common approach to fisheries management, the command and control (C&C) quota based regime. In these regimes governmental institutions create and validate knowledge which is used as input in decision-making. The setting of the TAC is the backbone of the system and attempts to reach long-term biological sustainability. However, these systems are very data hungry and also require ecological, economic and technical information to operate effectively, as well as information on compliance. Considerable institutional support, and cooperation by fishers is needed to attain this. Problems arise when rigid C&C ‘top-down’ decision-making process is backed by a biological oriented management research that does not take into account user’s participation and socio-economic knowledge. An interdisciplinary process of integrating, interpreting and communicating knowledge from several scientific disciplines must be complemented by and completed with user’s knowledge. Currently, new tools are being developed to improve management, and the authors highlight the use of Driver Pressure State Response (DPSR) indicators. Finally, the authors point out that the newly created Regional Advisory Councils (RACs), constitute the way that the CFP has chosen to improve management through the incorporation of users into the management process.

In Chapter 7, J. Rasmus Nielsen, Per J. Sparre, Holger Hovgaard, Hans Frost, and George Tserpes perform an extensive review of the knowledge-base issues around effort and capacity-control regimes. They provide case studies of the demersal gadoid fisheries of the Faeroe Islands, the Australian northern prawn mixed fisheries, mussel fisheries in Denmark, mixed hake fisheries in the Mediterranean and mixed demersal human-consumption fisheries in the North Sea. They find that knowledge base of an effort regulation system is more complex in multi-jurisdictional international fisheries management systems than in national and uni-jurisdictional systems, and in mixed fisheries systems compared to single species/stock fisheries. The definition of effort or fleet-based harvest control rules (HCR) provides the basis for any effort/fleet-based management regime. The HCRs must consider both the technical interactions between fleets and the biological interactions among the fish stocks. The knowledge basis for building up HCRs needs precise and quantitative definitions of fleets and fisheries, and a common unit of effort, in order to make it possible the allocation of effort and capacity between fleets
and countries. To ensure stability, there is a need for systems to monitor and tools to compensate for un-equal development in efficiency (effort creeping) over time between different national fleets and types of fleets. While for output management systems it is necessary to have relatively precise knowledge about resource abundance, effort-based management is less dependent on precise yearly stock assessment estimates and has a reduced need for annual assessment. It still needs reasonably precise stock assessment and forecasts to set an appropriate initial effort level and to respond to changes in efficiency. In respect to compliance and enforcement under effort regulation the authors suggest that control under this system is simpler than in an output regulation system because it is easier to observe activity of vessels than to monitor catch and output.

In Section 2, is our review of cross-cutting issues relevant to European fisheries management. As an orientation tool for readers who are not familiar with the main institutions involved in the fisheries management knowledge base in Europe, Troels Jacob Hegland provides us with an overview of them in Chapter 8. He describes a system that is currently undergoing reform and will most likely continue to evolve. While, it is difficult to predict the outcome of these reforms it is clear that there will not be a lesser need for qualified scientific advice.

In Chapter 9, Kepa Astorkiza, Ikerne Del Valle, Inma Astorkiza, Troels Jacob Hegland and Sean Pascoe examine the background and current issues with stakeholder participation in European fisheries management from the perspective of the knowledge base. They review the general literature on participation in policy making and locate the question of fisheries management within the wider context of the development of the European Union as a political project. Fishing and agriculture were the first sectors in which the responsibility for public policies was transferred from the Member States to the Union, and they have been regulated at Union level from its earliest stages, essentially in a top-down style. Legitimacy was always based on the fact that optimal policies could be developed, using totally objective criteria, in which neutrality had to be guaranteed by the impartiality of the procedures used in the process. Critics of this model emphasize an inclusive approach where social agents and stakeholders participate in the different levels of decision-making. Then from this perspective they examine the current development of the Regional Advisory Councils that are currently the main vehicle for stakeholder participation in the development of the Common Fisheries Policy. The authors see RACs as a single forum to develop a co-management model, at a larger scale than the local level and closer to the resource than the unique, but distant, European authority.

In Chapter 10, George Tserpes, Panagiota Peristeraki, and J. Rasmus Nielsen summarise the main ecological side effects of fishing and discuss various management approaches and actions for facing these effects. Due to the complex nature of the marine ecosystem and the lack of appropriate ecosystem monitoring schemes knowledge of the impacts of fishing on non-exploited species and the marine environment as a whole is fragmented. Clear and operational management objectives are critical for optimal management in relation to the ecologically sustainable exploitation of marine resources. However, consensus on those objectives between groups will be the major challenge for the implementation of an ecosystem approach within fisheries management. They examine efforts to develop Ecological Quality Objectives that could provide the basis for a knowledge base for
managing ecological side effects. However, they point out that the definition of ecosystem reference points and key indicators is at a very early stage.

In Chapter 11, Wim Demaré reviews issues around fisheries rather than stock based management. In Europe multiple fisheries and catching of multiple stocks as either target species or bycatch is a widespread issue. After providing an overview of fisheries-based management both the Mediterranean and the Atlantic he introduces the mixed-species TAC evaluation or MTAC model, a recently developed model for mixed-fisheries advice. Concerning data, although the needs are well-identified, the lack of discard data for most fisheries is a major weakness in the process. Improvements are expected, since the Data Collection Regulation (DCR) makes it obligatory to collect discard data for most fisheries. He shows that defining the linkage between associated stocks can become very complex, to the point that even if they were quantified the implementation of a management regime would be extremely difficult. It would also be hampered by the principle of relative stability that is used to guide the allocation of fish stocks between EU Member States.

In Chapter 12, Henrik Gislason reviews the requirements for the Ecosystem Approach to Fisheries Management (EAFM). It necessitates the development of scientific and institutional approaches for dealing with the inevitable increase in uncertainty without loss of legitimacy. Implementation of an ecosystem-based approach will probably generate a cry for additional monitoring, scientific investigations and complicated models and reference points, but as economic resources are limited, it is important to develop cost-effective performance indicators. Due to the limited understanding of how ecosystems respond to fishing, management institutions will have to adapt to considerable uncertainty. It is much more complicated to link cause and effect in ecosystem dynamics than in single species assessments, and often very difficult to conclude whether an observed change can be attributed to fishing or to natural environmental change. This is likely to pose new challenges for the process of scientific peer review and fuel discussions about the burden of proof and the practical implications of precaution. The chapter also reviews the use and further development of particular relevant management tools such as marine protected areas, gear modifications and eco-labelling.

In Chapter 13, Douglas Clyde Wilson and Sean Pascoe review the general problem of delivering complex scientific advice to multiple stakeholders. They examine some general issues such as the precautionary principle and the problems of dealing with multiple scales, resource uses and user groups, as well as the costs of data gathering and analysis for the development of advice. The use of indicators and formal models of economic and biological realities as the basis of cooperative fisheries management advice is discussed. They then apply the discussion to the development of management strategies that make use of harvest control rules. They conclude that a shift from an emphasis on objectivity to transparency in scientific advice can be facilitated by the use of formal models in participatory contexts.

In Chapter 14, Aaron Hatcher and Sean Pascoe examine the overall problem of compliance in fisheries management. They review the literature, including the various attempts that have been made to model compliance. Non-compliance affects the knowledge base through distorting the information used in fisheries stock and economic assessments. As a consequence, the controls imposed may not fully reflect the true underlying conditions in the fishery.
Section 3 contains two chapters in its review of scenario modelling and the way it can be used as support for fisheries management strategy evaluation. In Chapter 15, Jose De Oliveira, Laurence Kell, Andre Punt, Murdoch MacAllister and Sakari Kuikka review the use of operational management procedures (OMP) evaluation frameworks, which are simulation tools that allow management procedures to be developed in a precautionary manner. They review experiences with these tools in South Africa, Namibia, Australia, New Zealand, the United States and for the management of both the Icelandic and the North East Arctic cod stocks. The lessons they glean from this review include the need for data and analysis requirements to be realistic and well-specified, that the inevitable uncertainty should be explicitly taken into account, that management procedures should be rigorously tested with computer simulations, and that they must be used within an feedback monitoring system that allows the detection and response to changes.

In Chapter 16, Martin Aranda and Lorenzo Motos take on the question of management strategy evaluation (MSE). The MSE approach is wider in scope than the MP approach addressed in Chapter 15. Its purpose is to provide an objective basis for short or long-term decision-making. An essential component is the joint participation of managers, scientists, industry representatives and other stakeholders in data collection, assessment, advice and decision-making. The emphasis is on communication between the stakeholders. Presentation of simulation results is one of the means to enhance understanding of the performance of alternative strategies. Special effort should be placed in presenting the results of these simulations in an understandable way.

REFERENCES


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Section 1

GLOBAL EXPERIENCES WITH MANAGEMENT SYSTEMS RELEVANT TO EUROPE
Chapter 2

International Management of Shared Stocks

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2.1 INTRODUCTION

Since the rise of the Extended Fisheries Jurisdiction (EFJ) during the 1970s, the topic of shared stocks management has become an important matter of concern among policy makers, managers, stakeholders and researchers. The complexity of the management of such stocks lies in the diverse economic, biological and social implications that the exploitation of a resource by two or more participants brings about. The management of shared stocks is a complicated issue that can generate conflict, but, when addressed in a cooperative environment, it can bring about positive outcomes. Among these positive consequences are the development and improvement of research, which, in turn, generates a meaningful production of knowledge constituting the basis for a permanent improvement in the management of shared resources.

2.2 BASIC DEFINITIONS

John Gulland [1] was one of the first authors to state the scope of the term ‘shared stocks’. The author suggests that fishery stocks can be classified not only in terms of their biological conditions but also in terms of their relationship to national boundaries. According to this thought, Gulland classifies fishery stocks into five categories. The first group, which is not relevant for this review, is composed of stocks lying exclusively within the maritime boundaries of a single country. The other groups are:

(1) Stocks occurring within two or more Economic Exclusive Zones (hereafter EEZs), with movement across boundaries, but no clear migratory pattern.
(2) Stocks occurring within two or more EEZs with a clear pattern of movement (seasonal migration or movement according to developmental stages) between one zone and another.
(3) Stocks occurring wholly in the high seas beyond national jurisdiction.
(4) Stocks occurring both in the high seas and in one or more national jurisdictions.
Gulland makes a clear distinction within this classification. According to this, the first group is defined as a non-migratory group and stocks falling within are named trans-boundary stocks, while stocks within the remaining groups are defined as migratory stocks. Munro [2] refers to a definition by Caddy [3], who states that shared stocks can be defined as a group of commercially exploitable resources, distributed over or migrating across the maritime boundary between two or more national jurisdictions, or the maritime boundary of a national jurisdiction and the adjacent high seas, whose exploitation can only be managed effectively by cooperation between the states concerned. It is worth highlighting that an important concept for fisheries management arises in this definition: cooperation.

In the Food and Agriculture Organisation (FAO)-Norway Expert Consultation on the Management of Shared Stocks held in Bergen in 2002 [4], Munro defined the scope of ‘shared fishery stocks’ as:

(1) Fish resources crossing the exclusive economic zone (EEZ) boundary of one coastal state into the EEZ(s) of one, or more, other coastal states. These stocks are named transboundary stocks.
(2) Highly migratory fish stocks that, due to their highly migratory nature, are found within the coastal state EEZ and the adjacent high seas.
(3) All other fish stocks (with the exception of \textit{anadromus}/\textit{catadromus} stocks) found both within the coastal state EEZ and the adjacent high seas. These stocks are named straddling stocks.
(4) Fish stocks found exclusively in the high seas.

2.3 MANAGEMENT OBJECTIVES

The setting of achievable objectives is always a complicated task. Even within a country, clashing positions of involved parties turn this into a long and difficult process. In the case of shared stocks, the complexity of their management lies in the fact that a given resource is managed in order to be exploited by various states. Diverse and contrasting interests by the concerned parties arise here: conflicting positions are commonly found in the management of transboundary stocks but they are exacerbated in the cases of straddling and highly migratory stocks.

Even though there are some universal objectives in the management of fisheries, nations have different objectives. One country can aim, for instance, for a greater supply of a given resource, while another, which shares this resource with the latter, can aim for a smaller supply. This would determine a heavy unilateral exploitation of the resource. In these cases, an effective negotiation can bring about positive outcomes. The case of the Russian–Norwegian exploitation of cod in the Barents Sea is a good example of two countries that place a different value on the resource they share. In spite of this, they have managed to build an effective cooperative management [5]. On the other hand, bad negotiations or the absence of them have the potential to generate conflict. As a consequence, it is likely that the nation with a more developed fleet and processing infrastructure will be better off. Due to the contrasting objectives of fishing nations,
international and regional fisheries commissions have a hard task when setting objectives for the management of shared stocks. The difficulty here lies in setting objectives that can benefit all the concerned parties. For reaching these objectives, the setting of an effective policy that creates incentives for cooperation is necessary: a will to cooperate can soften the hard task of allocating resources between the concerned parties.

2.3.1 Objectives of states concerned

The good-will to cooperate for the sustainability of shared stocks may not be difficult to find among interested parties. This fact will help to set the general objective of resource sustainability. However, there are other contrasting objectives that are backed by the particular interests of the concerned parties. These objectives can range from increasing the local supply of protein as a staple food or raw material; increasing the supply of fish to foreign markets and, therefore, increasing revenue; the strategic development of fisheries and fleets, etc. Even though most of these objectives are common to the states, the degree of dependence on each of them varies according to the nation and, even in the case of neighbouring countries, the difference may be significant. The case of the western Africa countries Mauritania and Senegal, and their common exploitation of sardinella, is a good example. In Senegal, the exploitation of the resource is done by the artisanal fleet; whereas, in Mauritania the exploitation is done by the industrial fleet that supplies raw material to fish meal plants [4].

Gulland [1], focusing on the objectives of transboundary stocks management, considers two groups:

1. The countries concerned have similar economies and ultimate objectives.
2. The countries involved have different economic and social conditions as well as different management objectives.

The first is the simplest case in managing shared stocks. Here, states will probably have similar strategic and tactical objectives: for example, to establish the total allowable catch (TAC) and the procedures to reach it. In this case, the problems that arise are those related to technical methods [1]. The second case, which is the most common case in management, can be understood with a simple example from Munro [6]. In this hypothetical case, two coastal states share a certain resource. On the one hand, one state might favour high long-run TACs, and accept low catch rates. On the other hand, the other state might favour low long-run TACs, but high catch rates. In this case, the problems that arise are those related to finding the means to develop a mutually acceptable management programme.

2.3.2 Conflict and other problems

Conflict means that people have or see themselves as having incompatible goals. This fact is valid for local as well as international scenarios. Management of shared stocks is a fertile field of controversy. Controversies are exacerbated by inconsistent regulations, by weakly defined rights or even by an absence of regulations and agreements. It is noticeable
that cases of controversy arise between the distant water fishery nations (DWFNs) and the coastal states. For example, a controversy arises when coastal states claim the right to exploit and safeguard straddling resources outside their EEZ. On the other hand, DWFNs claim the right to freely fish in the high seas. According to Munro [6], the origin for such controversy is the lack of clarity regarding rights in these zones. It has brought about several conflicts between concerned nations and it ensures non-cooperative management.

Similar controversies are found in the case of resources occurring in the EEZ of a given state. This problem is exacerbated by the lack of clarity of international regulations – it was one of the initial sources of controversy relating to the Law of the Sea Convention which, in Part V, provides the basic rules for international fisheries management. Article 56 of the Convention provides coastal states with sovereignty over the natural resources occurring within EEZs. This Article states that, within the 200 mile EEZ, coastal states enjoy sovereign rights for the purpose of exploiting, managing and conserving resources [7]. It being understood that this sovereign right includes all resources available in the EEZs waters, this provision creates great controversy because it includes resources that are also managed and exploited by other nations as transboundary, straddling and highly migratory. Moreover, Article 64 provides two paragraphs that are seen as contradictory [8]. The first paragraph states that where no international management body is established the relevant coastal states should cooperate with other states that harvest the resources. From this paragraph, it is clearly understood that DWFNs are empowered with a role in the management of these resources. And yet, the second paragraph is to be applied in addition to other provisions of Part V, which can be understood as ratifying the empowerment of coastal states with full sovereignty. Thus, inconsistency in regulation provides conflicting parties with legitimate means to litigate against each other.

2.4 MANAGEMENT TOOLS

Management of shared stocks is a formidable task due to its high complexity: it tries to coordinate the efforts of two or more states which, in most cases, have different goals, in order to carry out sustainable exploitation. Two approaches characterise the shared exploitation of a resource: cooperative and non-cooperative management.

2.4.1 Cooperative and non-cooperative management

Because of the complexity and costs of achieving mutually acceptable management actions, cooperative management provides a means to achieve those actions. Various authors highlight the importance of this key tool. Churchill [9] considers cooperation over the management and conservation of transboundary fish stocks as desirable in many cases.

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1 There is a tendency among coastal states to claim rights for exploiting and managing resources that straddle EEZ waters and high seas. Coastal States categorise this right as the consistency principle while DWFN categorise this as creeping jurisdictionalism [8]. A particular case of a claim to exploit and manage not only straddle resources, but also non living resources, beyond the national EEZ is seen in the Chilean proclamation of the Mar Presencial [52].
Shared stocks management

Hanneson [10] states that management of stocks that migrate across national boundaries requires cooperation between the countries involved. Munro [2, 6] argues that cooperative management is important for the long-term sustainability of most resources.

Munro [2] points out that it is dangerous to assume that non-cooperative management of shared stocks will lead to adequate management programmes. Regarding non-cooperative management, the FAO [4] refers to a paper by Willmann in which the author states that, according to past evidence in various fisheries around the world, shared stocks managed in absence of cooperation are doomed to become overexploited with potentially dramatic economic, social and nutritional impacts on the countries concerned. The author highlights that this fact is in conformity with game theory, specifically with the expected outcome of the ‘Prisoners’ Dilemma’.

According to Munro et al. [11] the most vivid example of the consequences of non-cooperative management is the case of the joint exploitation of Pacific salmon by the USA and Canada. Another relevant case of non-cooperative management is the fishery of the Alaskan pollock in the ‘donut hole’ of the Bering Sea. In this case, non-cooperative management between the coastal states concerned, the USA and Russia and the DWFNs – Japan, China, Poland and the Republic of Korea – allowed a heavy overexploitation of these straddling stocks by the DWFNs. Unfortunately, when cooperation was finally reached between 1991 and 1994, the damage had already been done [12].

Caddy [3] provides another example of non-cooperation: the case of the management of straddling stocks in the Adriatic Sea. According to this author, the stocks’ level recovered as a result of the conflict in former Yugoslavia. However, non-cooperative management generated a rapid increase in fleet capacity, which was a threat to the healthy levels of the stocks. Even though Caddy reports that cooperation in research was carried out, he suggests that this was not sufficient to face the emerging threats [13].

Cooperative management is the result of a process that usually begins with cooperation in research and data gathering and exchange. Regarding this process, Gulland [1] recognises two levels of cooperation:

(1) First level: Cooperation in research alone, without reference to coordinated management programmes. This is likely to be the most palatable type of cooperation for the parties involved, since the participants would benefit from the exchange of information and data. However, it is possible that one of the parties may suspect that the information it shares may benefit the partner’s exploitation of the resource. Munro [6] suggests that if, for this or for other reasons, it were not possible to achieve cooperation in research it would also not be possible to achieve cooperation in active management of the resource.

(2) Second level: This is active management, which involves the establishment of coordinated management programmes. Gulland [6] points out that it requires:

(a) Determination of an optimal management strategy including, *inter alia*, the determination of global harvest over time.

(b) Allocation of harvest shares among the participants.

(c) Implementation and enforcement of coordinated management agreements.
Garcia [14] also recognises the need to develop and implement joint management, taking cooperation in research and data gathering as a basis. According to this argument, the first level of cooperation leads to cooperative management through technical measures, in a first stage, and through agreements regarding access to and allocation of resources, in a second stage. This provides the basis for bargaining over management strategies and harmonised regulation. The conclusion of agreements over surveillance and control complete the process. Garcia adds that effective management of shared stocks requires negotiation and agreement between the states concerned in order to settle some key issues such as resource access allocation or application of compatible management measures.

FAO [4] refers to a list of requirements to achieve stable cooperative management arrangements. This list, elaborated by Willmann, specifies that:

1. Every country should be better off with cooperation than without it.
2. The outcome of cooperation should be perceived as fair and equitable.
3. The above conditions need to be held over time (time consistency).

Willmann adds a list of the major benefits to be expected from the cooperative exploitation of small pelagic resources. This list was originally made to show the benefits to be gained from the cooperative exploitation of pelagic resources off the Northwest coast of Africa2, but it can be readily applicable to a wider range of resources:

1. The optimisation of the economic, social and nutritional contribution of resources to the country and people of a given region.
2. The avoidance of the collapse or depletion of the small pelagic resources.
3. The improvement of knowledge about the resources and their associated ecosystem.
4. Harvesting opportunities for the national fleet within another country’s EEZ.
5. Harvesting opportunities for all fleets throughout the EEZ of participating countries.

Another example of cooperative management is the case of the Norwegian spring spawning herring. This stock, which is both a transboundary and a straddling stock, migrates from the Norwegian EEZ (after spawning) through the EEZs of the European Union (EU), the Faeroe Islands and Iceland. The resource collapsed in the late nineteen-sixties and took twenty years to recover. The moratorium that was declared in 1969 was lifted in the

2 Not all coastal states have the technology and expertise to exploit a potential resource. In these cases, an accurate agreement in which the coastal state makes an agreement with a DWFN with more expertise and technology can be highly beneficial for both parties. Among the various benefits for the coastal state are agreed supplies of fish for both direct consumers and factories, training of local fishermen and side-payments. In order to avoid misreporting and other non-compliance behaviour, the agreement must be transparent and enforceable. In spite of this, the implementation of an Monitoring, Control and Surveillance (MCS) system can be difficult and expensive.

3 Side payments or transfer payments are benefits from the shared resources that are not derived from the harvesting activities of a state’s fleet within its own EEZ, but through beneficial monetary or ‘in-kind’ transfers to the country from one or more countries. According to Munro [45], side payments ease the resolution of resource management conflicts. The resource is managed as if it were managed and exploited by a sole owner.
late nineteen-eighties. After recognising that cooperative management was necessary for maintaining the healthy level of the stock, Norway, Iceland, Russia, the Faeroe Islands and the EU established a cooperative arrangement in the mid 1990s. The arrangement has shown signs of stability and is considered successful. However, due to the migratory pattern of the resource it is considered that allocations will require renegotiations [2].

2.4.2 Negotiation and arrangements

Arrangements and agreements are fundamental to the development of appropriate management of shared stocks. According to Gulland [1], two types of arrangements are possible. In the first case, a permanent commission is established whose recommendations are binding on member countries. In the other case, agreements are made directly between governments, possibly on an ad hoc basis for each year. Despite this classification, the author highlights that, in practice, a hybrid procedure has sometimes been followed. Churchill [9], focusing on transboundary stocks, recognises four types of arrangements, which differ in form:

(1) Those concluded under a pre-existing framework agreement (for example, Norway–EU in the North Sea; Norway–EU–Sweden in Skagerrak–Kattegat).
(2) Those where measures for the management of transboundary stocks are taken by a bilateral commission as for Iceland–Norway (North Sea herring) and Argentina–Uruguay (hake).
(3) Those where measures are adopted by a regional fisheries commission, which is appropriate in enclosed or semi-enclosed seas where fish migrate between the zones of several states. This is the case for the exploitation of salmon, cod, herring, and sprat in the Baltic Sea, which is regulated by the International Baltic Fisheries Commission.
(4) Those where parties undertake in a general way to cooperate over the management of transboundary fish stocks on an ad hoc basis, but where no detailed arrangement have yet been adopted under these provisions (for example, the Gulf of Guinea; and conservation cooperation among the states bordering the Atlantic Ocean).

According to Garcia [14], states may generically wish to negotiate an agreement specifying matters such as cooperation in research and stock assessment. At a later stage, the states may wish to implement an agreement with precise issues, such as joint training of personnel in standard management procedures, common marking of vessels, use of radio call signs and the exchange of registers of authorised vessels. The exchange of this information could lead to defining procedures for enforcement.

A good example of a cooperative agreement is the one reached by the nations fishing Alaskan pollock in the ‘donut hole’ of the Bering Sea. In this case, all the countries concerned, pushed by the evident collapse of the stock, conducted a series of conference meetings between 1991 and 1994, which resulted in the signing of an agreement in 1995. According to Dunlap [15], the agreement provided a unique combination of enforcement mechanisms, which offered the potential to become one of the most effective agreements
ever reached. Under the terms of this agreement the contracting parties agreed to joint efforts for the conservation, management, and optimal utilisation of the pollock stocks in the ‘donut hole’. Among the most important aspects of the agreement are [12]:

1. Provisions for the determination of annual harvest levels and individual nation quotas for each year.
2. Effective mechanisms for dealing with non-complying parties.
3. Broad provisions for dealing with nations who are not party to the agreement and intend to undermine the objectives of the Conference.
4. Cooperation on research and exchange of fisheries data.
5. Satellite tracking for all fishing vessels.
6. Establishment of a scientific observer programme for full coverage of fishing activities.

One of the most important factors to be taken into account when designing agreements is duration. According to Queirolo et al. [16], an agreement that guarantees contracts or investments for long-term access to international markets can be highly beneficial for the coastal nation, while steady supplies of raw material would motivate the foreign partner. Sufficient duration imparts an effective ‘property interest’ in the resource. Consider the case of a DWFN seeking to fish in a given EEZ: in the case of limited duration, the ‘property interest’ of the contracting party is so reduced that it will always seek to accelerate the harvest to the point of biological overfishing; a short lifespan agreement has the potential to accelerate harvesting towards the end of the agreement’s duration in such a way that increases costs and reduces conservation measures. This may occur even with intensive monitoring and extensive enforcement [16].

2.4.3 Allocation

Allocation of shared stocks is likely the most complex issue in management of shared stocks. It requires taking into account economic, social, technological and biological factors that are to be considered in political negotiation. However, it has received little attention in technical literature [3]. According to Garcia [14], allocation mechanisms and the negotiation of specific shares need to be determined directly by the parties with clear rules established through negotiation, if necessary with the support of an independent arbiter. Being a gradual process, allocation involves solving two key problems [1]:

1. How big a share should each participant get?
2. How should the shares be measured?

According to Gulland [1], a state is likely to expect a large allocation when the productivity of the stock is critically dependent on conditions in its EEZ, especially when its activities can affect those conditions. For example, stocks which have nurseries in inshore areas that can be damaged by cutting of mangroves. If a state controls these activities for the benefit of the fishery, it would expect a large allocation. Gulland points out that there are other non-biological factors that need to be considered in the process of allocation: factors
as diverse as the value countries set on being able to take a catch because of possessing a highly efficient fleet, or having a good marketing and distribution system; or other factors such as high requirement of protein or lack of alternative labour opportunities for those currently involved in the fishery. Gulland [1] also states that the ‘value’ a given country places on the exploitation of a certain stock is not easy to measure in comparable terms. For instance, an underdeveloped country places a high value for the fish because of a high local requirement of protein and not due to its monetary value, which can be low. However, these economic considerations do not give quantitative guidance on the magnitude of the allocations. In spite of that, appropriate limits to the possible allocation will be given by the benefits that each country would obtain in the absence of agreement on allocation.

Regarding the second problem, Gulland [1] suggests that measuring the stocks’ shares as a certain tonnage out of a given TAC is often the best choice because of the clarity of the units. However, it is necessary to point out that the setting of an accurate catch limit is difficult in cases of stocks that show great fluctuations. The author considers relative shares as less clear: for instance, an agreement that allows one state to operate 75 medium size trawlers and another to fish with 25,000 artisanal fishermen. Moreover, the problems in the latter case will be increased when one state decides to improve the operating efficiency of the fishers. What adjustment should be made to the number of units allowed to operate will be the cause for disagreement and renewed conflict. It is likely that agreements on the allocation of shared stocks will continue to be expressed in terms of a share of a TAC. However, consideration should be given to other approaches: for instance, fishing effort expressed in number of vessels [1]. Regarding this last suggestion, it is necessary to highlight that allocation in terms of fishing effort will be a matter of continuous adjustment because of technological creep and capital stuffing4.

FAO [4] recognises that one of the most critical and difficult decisions to agree on is the selection of the criteria to be employed in the allocation of harvesting opportunities (catch, effort or capacity quotas) among the countries involved. FAO sums up that this selection could be based on:

1. Distribution of the harvestable part of the stocks over the EEZs of the participating states.
2. Historical catches in the EEZ of the participant states.
3. Historical level of deployment/size of harvesting capacities in the EEZs of the participating states.
4. Various other factors such as, for example, the contribution of catches to domestic food supply or the number of people engaged in the fisheries.
5. A combination of the above criteria with certain weights attached to each of them.

4Capital stuffing is a way to circumvent regulations which aim to restrict fishing effort through investment in technological improvement or input substitution. This brings about an increase in efficiency which is always difficult to assess.
2.4.4 Management instruments

The chief document for management of shared stocks is the United Nations Convention on the Law of Sea of 1982 (hereafter the Convention), which entered into force in 1994. The Convention sets up the legal framework for all subsequent international arrangement and agreement relations to the use of the oceans and seas. Arising from the Convention, and designed to strengthen its provisions relating to high seas fisheries and straddling stocks, are the United National Fish Stocks Agreements and the FAO Compliance Agreement [17]. Van Houtte [18] points out that the Convention exhorts states to cooperate or to negotiate in order to address shared stocks management issues and the conservation of highly migratory stocks. Cooperation as a basis for shared stocks management is an important issue in the Convention. Article 61.5 advocates cooperation in research through exchange of relevant and available data. In this Article, the Convention states the obligation to contribute and exchange available and relevant data. However, FAO [4] states that the obligation to collect the necessary data is not elaborated in the Convention. Nor does the Convention specify what kind of data should be included under the obligation, other than references to ‘scientific information’, ‘catch and effort statistics’ and ‘other data’. The FAO recommends that agreements on conservation and management of shared stocks should develop further rules regarding the duty of the parties to collect and exchange relevant data. The FAO adds that the FAO Code of Conduct for Responsible Fisheries gives more extensive provisions on data gathering and exchange than the Convention.

In two key articles (63.1 and 63.2), relevant states are admonished to cooperate. Article 63.1, which is the transboundary stocks paragraph, reads:

Where the same stock or stocks of associated species occur within the exclusive economic zones of two or more coastal states, these states shall seek, either directly or through appropriate sub regional or regional organisations, to agree upon measures necessary to coordinate and ensure the conservation and development of such stocks without prejudice to other provisions of this part.

Article 63.2, which is the straddling stocks paragraph, reads:

Where the same stock or stocks of associated species occur both within the exclusive economic zones and in an area beyond and adjacent to the zone, the coastal state, and the states fishing for such stocks in the adjacent area shall seek, either directly or through appropriate sub regional or regional organisations, to agree upon measures necessary for the conservation of these stocks in the adjacent area.

Van Houtte [18] comments that, in Article 63.1, the Convention imposes a duty to negotiate arrangements but there is no duty to reach an agreement. Moreover, the Convention does not further elaborate on the management and conservation objectives or on the allocation of catches among the states concerned. Regarding Article 63.2, Van Houtte states that this Article provides an essential starting point for the resolution of problems that have arisen in the implementation of straddling stock regimes. However, the author comments that this provision does not offer much guidance as to how problems involved in managing straddling stocks can be addressed. Moreover, cooperation is called
for taking measures for conservation purposes regarding high seas (adjacent zones) but not regarding EEZs. In this context, there are two contrasting positions: the first is the position of the DWFNs that claim the right of fishing in the high seas, which includes areas adjacent to EEZs; the second is the position of the coastal states, which claim the right to protect the resources being exploited within their EEZs and which are affected by the exploitation in neighbouring areas outside their jurisdiction. These two positions are difficult to conciliate, especially because DWFNs see the position of the coastal state as a will to extend their jurisdiction beyond their EEZ, this is known in legal jargon as creeping jurisdictionalism.

Article 64 of the FAO’s Code of Conduct for Responsible Fisheries deals with highly migratory species. This article suggests that the coastal state and other involved states shall cooperate either directly or through international organisations in the conservation and optimal utilisation of the species within and beyond the EEZs. This has been considered to constrain the coastal state’s management of highly migratory species when they straddle into their EEZs [11].

Another important legal instrument is the 1995 UN Fish Stocks Agreement (further called the Agreement). In August 1995, the UN Conference on Environment and Development adopted the Agreement for the conservation and management of straddling fish stocks and highly migratory fish stocks. This agreement was created with the general objective of overcoming the fisheries management problems associated with the failure of the UN Convention. However, Tahindro [19] affirms the Agreement is in accordance with the relevant provisions of the Convention. Juda [20] and Tahindro [19], among other authors, state that the new Agreement incorporates the concept of precaution: that is, states are to apply a precautionary approach widely to conservation, management and exploitation of stocks, and to use caution when information is unreliable. The authors also maintain that the Agreement strengthens the role of regional or sub-regional fisheries management organisations, through which states will establish a cooperative context. states assume a number of obligations within such a cooperative context: *inter alia*, to obtain and evaluate scientific data; implement and enforce conservation and management efforts through effective monitoring; and control and surveillance. Finally, the authors emphasise that the Agreement promotes ecosystem-based management. The Agreement not only creates a detailed framework for the management of these stocks, but also integrates management and conservation within the context of the need to avoid adverse impacts on the environment, the preservation of marine biodiversity and the integrity of marine ecosystem.

Van Houtte [18] adds that the 1995 UN Fish Stock Agreement implements the Convention and provides more detailed provisions for the management of straddling stocks and highly migratory stocks. The author highlights that the Agreement not only creates a detailed framework for the management of these stocks but also integrates management and conservation within the context of the need to avoid adverse impacts on the environment, and of the importance of the preservation of marine biodiversity and the integrity of marine ecosystem. The Agreement deals with issues such as the duty to cooperate, the issue of compatibility and management measures, mechanisms for cooperation, duties of flag states and compliance and enforcement.
2.5 THE PRODUCTION OF KNOWLEDGE

A precondition for reliable quota management of living resources is that sufficient knowledge about the resources has to be available [25]. In this context, knowledge production and use is essential to management of natural resources. Caldwell points out that knowledge is derived from information, and, consequently, the collecting, organising, analysing and testing of information becomes an essential element of public policy administration [21]. It is essential to highlight that not only biological knowledge about the resources is required; in this context, Jentoft and McCay [22] point out that knowledge production needed for effective management must be ‘multi disciplinary’. In other words, biological data analysis must be supplemented with socio-economic data analysis. Additionally, Berkes et al. [23] suggest that fisheries management also requires information of various types coming from a variety of sources: such as, traditional, local and administrative. According to Charles [24], the importance of knowledge production and use is determinant in accurate management of fisheries resources – there may be a correlation between the collapse of several stocks and the state of the knowledge base that was used in decision-making. He suggests that one of the contributors to fisheries collapse may have been the combination of a lack of knowledge and a failure to use all available sources of information and knowledge at that moment.

The obligation of states to exchange information is clearly stated in the UN Law of the Sea Convention. In its Article 61.5, the Convention calls states to contribute and exchange available and relevant data. However, FAO [4] considers that the Convention does not give more precise detail on what kind of data should be included under the obligation, other than the reference to scientific information, catch and effort and other data. FAO [4] states that the voluntary Code of Conduct for Responsible Fisheries has more extensive provisions on information gathering and exchange than the Convention, and can encourage countries to exchange information on shared stocks. In this regard, the Code of Conduct in its paragraphs 6.4 reads ‘Conservation and management decisions for fisheries should be based on the best scientific evidence available, also taking into account traditional knowledge of the resources and their habitat, as well as relevant environmental, economic and social factors.’

There is an increasing trend among scientists and managers to incorporate both traditional knowledge and industry’s knowledge in the diverse stages of the management process. These stages include data collection, assessment, advice, decision-making, monitoring, control and surveillance (MCS) and enforceability. The role of resource users as both suppliers of knowledge and active participants is an essential component of several of the most advanced management systems in the world (examples include Australia, South Africa, Canada, and the USA). Despite the fact that some success has been achieved in incorporating users’ knowledge in modern management, the incorporation of this knowledge into modelling remains a major challenge to scientists. In the particular case of shared stocks, fishers’ knowledge regarding migratory patterns and the interrelation between species and the ecosystem is an important input that helps decision-makers to allocate resources among parties, decide fishing seasons and assess the impact that a given exploitation will generate on the resources on zones adjacent to state boundaries. Willmann [4] highlights the view that joint exploitation of resources generates as a positive
outcome a meaningful improvement on knowledge. In this case, cooperative management seems to be the best means to produce knowledge. Further, even though information can be produced without cooperation, McKelvey has found out that increasing the demand for information without cooperation can be in certain circumstances be damaging [6]. In this case, competition may push states to avoid information exchange because of the fear of free-riding.

2.5.1 Cooperative research or the first level of cooperation

Cooperation in shared stocks management often begins with collaboration in research and data gathering and exchange. Gulland [1] names this initial process the first level of cooperation. According to Gulland, this type of cooperation is likely to be the most acceptable to the parties involved, since they would benefit from the exchange of information and data. However, this first level of cooperation could also bring some negative outcomes. It is possible, for instance, that one of the parties may suspect that the information it shares may benefit their partner’s exploitation of the resource more. Moreover, a greater production of information by one of the parties could be used as a tool to get better advantages from agreements. Despite these possible negative outcomes, Munro [6] stresses that the importance of cooperative research is so fundamental that, whether for this or other reasons, if it were not possible to achieve cooperation at this stage it would not be possible to achieve cooperation in active management of the resource. Munro comments that, in the FAO–Norway expert consultation on the management of shared stocks, several cases presented by participants from all over the world described the significance of scientific research. The author highlights that, even in the cases where active management is not currently carried out, cooperative research is relevant as a starting point for joint management in the future. These are the cases of, inter alia, the States of the South East Pacific (Ecuador, Peru, and Chile) and Northwest Africa. On the other hand, during the same expert consultation it was demonstrated that problems such as a lack of funding affect effective scientific cooperation, which in turn prevents effective cooperative management: this is the case in Argentina and Uruguay. The author also refers to the case in which a lack of scientific knowledge about a resource (South African hake) impedes effective cooperative management between South Africa, Namibia and Angola.

2.5.1.1 Joint research as a first step of producing knowledge and active management

A good example of collaboration in research is the case of the Russian–Norwegian management of resources in the Barents Sea. It started in the early 1950s with non–governmental cooperation between research institutes, and is acknowledged as one of the longest and best traditions in the Norwegian–Russian collaboration on fisheries management. Research is carried out by research institutions in both countries. Data is gathered from both commercial landings and scientific surveys. Scientific advice is taken from ICES but decisions are taken by the bilateral Commission on an annual basis. Both parties are currently working on the simplification of procedures, which will ease
the entry of research vessels into each other’s EEZs [25]. Furthermore, cooperation not only entails joint research, but also information exchange in support of enforcement of regulations. In this context, the parties also agree to exchange information on involved vessels, ship-owning company quotas, satellite tracking data, transhipment data and other information [26].

Having this basis of joint research, joint management of cod by Norway and Russia is a successful example of cooperative management [5, 27]. An agreement between these two nations has been in place for the last three decades. The agreement considers fixed percentage allocations of the TAC to cod and haddock. Another characteristic of the agreement is that, due to biological characteristics of the cod stock, neither participant is restricted to harvesting their respective quota in their respective EEZs. In fact, Russia has fished more cod in the Norwegian EEZ because cod reaches the adult stage in Norwegian waters. This case is also remarkable for the use of side payments which secure flexibility. Finally, this case highlights the different value that nations place on a given resource. Traditionally, cod is more valued by Norwegians than by Russians. In this context, Norway gets bigger shares of the global quota (even though the initial allocation was a 50–50 sharing); in return for this greater share, Norway gives quotas to Russia for them to harvest other resources.

2.5.1.2 Scientific cooperation as a means of creating knowledge and facing threats

Scientific cooperation can also be a means of both creating knowledge and dealing with threats. The Commission for the Conservation of the Southern Bluefin Tuna (CCSBT) is an example of the benefits to be obtained from cooperation. The role of science in the management of southern bluefin tuna is crucial to establishing the TAC, but it also provides partners with scientific material to respond to external threats. In the early 1990s, there was some pressure for declaring bluefin tunas as endangered species. The proposal was issued by Sweden and proposed to list northern bluefin tuna as an endangered species on the Convention on International Trade in Endangered Species—(CITES). This proposal generated concern in Japan, New Zealand and Australia, since the inclusion of northern bluefin tuna would have meant pressure by conservationist groups to have southern bluefin tuna included as well. This threat pressed the parties already cooperating within the Convention for the Conservation of the Southern Bluefin Tuna to conclude agreements on conservations issues which had been pending since 1984 [28].

2.5.1.3 The role of symmetry of knowledge tenure in cooperative management

Gulland [1] suggests that, despite its potentially positive outcomes, cooperation in research can influence harvest allocation and, therefore, become a ‘tool of combat’ in negotiation between the states involved. The rationale is that there is a possibility that the partners hold information asymmetrically. The greater knowledge in the hands of one partner would become a tool for that state to obtain benefits. This is likely to happen when an arrangement is made between a DWFN and a coastal state. Since distant water fishing fleets (DWFFs) usually belong to highly industrialised countries with expertise in distant fishing, processing technology and a presence in international fish markets, they may know more about the condition of the resource, its population dynamics, its exploitation
and its trading. Because most of their activities are not easily tracked by the coastal state, information on catch, discards, and other variables, crucial to economic decisions, can be used for the benefit of the DWFN [16]. Thébaud [29] suggests that a means to minimise the information asymmetry between the parties is to establish an institutional framework that will not only reduce this asymmetry but will also minimise the costs of monitoring and enforcement.

McKelvey and Golubtsov [30] analyse the expected outcome of two scenarios under different information structures: one under symmetric, and the other asymmetric, conditions. Both situations relate to knowledge of the bio-economic parameters involved in the stock exploitation. One of the main conclusions posited by the authors was that, under fully symmetric information conditions, a cooperative arrangement could assure higher returns than a competitive one (this scenario included low harvesting costs and highly precise information), while a competitive harvest could tend to deplete the resource. On the other hand, under asymmetric information, the competitive situation becomes better for the party who has greater precision of information (who will increase their harvest rate). This player has no incentive to share information and thus to establish a cooperative agreement – for example, by adopting a common exploitation policy.

2.5.2 Non-cooperative management and the fear of knowledge sharing

The joint exploitation of Pacific salmon by the USA and Canada is a classic example of non-cooperative research and management. This case is analysed by Miller [31]. Pacific salmon are found between northern California and Alaska. They are exploited mainly in the mouths of the rivers when they are returning to spawn and die. Pacific salmon is considered both a transboundary and straddling species. During a certain stage, efforts to use cooperative management were made between the parties, which produced positive outcomes for both the USA and Canada, as has the inclusion of Article 66 (the anadromus species article) in the United Nations Convention of the Law of the Sea. This Article states that direct fishing of salmon in the high seas is contrary to international law. In spite of the proven benefits of cooperative management in the past only the Fraser River Pacific salmon stock was managed cooperatively; for the other stocks non-cooperative management was predominant. Even though both the USA and Canada possessed the means for enhancing the stock in their respective rivers, due to the competitive nature of the fishery, they were not willing to initiate stock-enhancing cooperative projects due to the fear of free-riding. After several years of negotiation, the two parties signed a treaty in 1985. However, the treaty faced several difficulties and was finally abandoned. The countries again chose competitive behaviour, which brought about the expected degradation of the resources. Finally in 1999, the countries signed an agreement for cooperative management to counteract the negative outcomes.

2.5.3 Biological and ecological knowledge required

2.5.3.1 Biological knowledge

Joint management of international fish stocks requires thorough knowledge of stock distribution, migration patterns and stock structure. This knowledge is essential to back
up the accurate allocation of resources. In this context, cooperation in research is essential to gather the information required to manage shared stocks [1]. Data gathering faces several problems derived from the difficulty of tracking changes in migration patterns, seasonal and spatial distribution of stocks, and holding information on different patterns of exploitation in each of the sharing countries. Holding this kind of information influences economic aspects of negotiations, since the geographical and seasonal changes in the availability, as well as in the quality, of the fish may directly influence both the cost and value of the catches. The influence of distribution and migration patterns not only applies to problems of competition and conflict between sharing countries, but also to competition between national fleet segments or national fishermen groups [32].

Essentially, data-sets for effective shared stocks’ assessment and management should comprise summaries of total catches broken down by categories such as area, gear, and the size and age of fish [1]. Abundance indices, such as CPUE of specific fleet segments, usually selected on the basis of gear and spatial range of activity, also have to be included in the data set to be used as an input in assessments. Most analytical assessments of Northern Atlantic stocks, for example, imply length or age structured models (Virtual Population Analysis), which require that the length and age compositions of the catches is estimated. Depending on the assumptions of the model to be used, it is necessary to estimate effective effort, for example, effort targeted at different species (mixed fisheries), or to standardise effort to remove the effect produced by changes in efficiency or in tactics overtime. With regard to data from scientific surveys: indicators coming from trawl surveys, acoustic surveys and egg surveys are commonly used to provide input data for assessments. It is also necessary for certain biological data to be collected for assessments: these include growth and weight at age data and data on maturity. The data-set should also comprise data processed by research institutes which provide corrections for misreporting, and discards data collected on fishing vessels by scientific observers. It is necessary to highlight that when scientific surveys are not carried out on research vessels they should apply a statistically valid protocol [33].

According to Charles [24], research surveys may produce a temporally static picture of the status of stocks and may cover only a fraction of the fishable area, thus failing to capture the spatial diversity of stocks. This can lead to possible biased estimates of the stock, depending on stock structure and migratory patterns. Although detailed information on the amount and composition of commercial catches is crucial for the quality of stock assessment, catch reports may have substantial errors (the actual amount of fish caught is unknown), and this can seriously affect the reliability of the scientific advice on the future development of stocks and catches [34]. This problem is particularly acute in the case of shared stocks since it is always a hard task to gather catch information from fleets of different nationalities. This is particularly the case given that data is also biased by illegal practises, which are easily carried out beyond the scope of the national MCS.

2.5.3.2 Ecological knowledge

According to Cochrane [17], the inclusion of ecological data in fisheries management is recognised to be essential for the sustainable and efficient use of resources. Since target
species are dependant for their survival and productivity on the ecosystem in which they live, any ecological change will impact on target resources. The manager needs to be aware of those changes, whether natural or caused by fishing or other human activities (examples include the impact of trawlers on the seabed, or the impact of discards). Such awareness enables the assessment of the possible impact of the changes and the strategy to be adopted in order to minimise damage to the ecosystem [17]. Scientific understanding of ecosystem interactions and dynamics is still very limited and there is therefore a high degree of uncertainty in the prediction of ecosystem behaviour. Among the data required to provide information to fisheries managers are: total catches of bycatch species or indicator species and estimations of discarding per fleet on an annual basis; length and age composition of bycatch species or indicator species; impact of fishing gear and activities on the physical habitat; and changes in habitats caused by other human activities [17].

In the context of joint exploitation of resources, ecological factors have to be taken into account when allowing DWFN to harvest fish in national waters. Large fishing fleets may use trawlers, which both impact on the sea bed and also generate large quantities of bycatch that may be discarded. Moreover, targeting species not included in the agreements, fishing of undersized specimens and misreporting of catches are among other common problems. Even though these problems are widely spread, the lack of accurate MCS systems for DWFFs may worsen the situation.

2.5.4 Economic knowledge and costs of cooperation

In managing shared stocks, managers should have a comprehensive knowledge of the underlying economic factors behind the participant’s behaviour under the different scenarios—transboundary stocks, straddling stocks and highly migratory stocks. Fortunately, the production of knowledge on the economics of shared stocks is especially meaningful in the case of transboundary stocks and it is possible to explain and predict, through theoretical bio-economic modelling and the empirical application of models, the behaviour of a fishery exploited by two or more participants. This knowledge has mostly been produced in academic quarters after the rise of the extended fisheries jurisdiction. Munro [6] highlights the contribution of economists to our understanding of the dynamics of shared stocks’ management, which has impacted favourably on the works of specialists in various disciplines as diverse as marine biology, policy-making and law.

Most of the first studies related to transboundary management and have as a basis a dynamic model of fisheries developed by Clark and Munro [35], which was in turn based on the well-known Schaefer model [36] and Nash’s theory of two-person co-operative or non-co-operative games [37, 38]. The study of the economics of shared stocks has been mainly developed by Munro [39–41] who investigates the optimal management of resources owned by two states through inclusion of conflicts in the management strategies of both states. Those conflicts arise from differences in perceptions of the social discount rate, the fishing effort costs and consumer preferences. Munro assumes one of the states to be more conservationist than the other and to have a greater incentive to invest in the resource.

A variety of works warn managers about the deleterious outcomes of non-cooperative management: inter alia Clark [42] and Fisher and Mirman [43, 44]. In contrast, other
works, such as the empirical analysis of the Arctic-Norwegian cod fishery by Armstrong and Flåten \cite{5} demonstrates the significant gains of the actors involved in cooperative management. Munro’s \cite{45} analysis of the Pacific salmon fishery, shared by the USA and Canada, shows how conflicting objectives and non-binding agreements usually result in overfishing. Research on the economics of straddling and highly migratory stocks \cite{8, 46, 47} offer a picture of the interrelations between the DWFNs and coastal states. Findings suggest that DWFNs, which bear higher operating costs, should be compensated by coastal states through *side payments*. In this case, it is remarkable how theory helps managers to set an optimal policy.

The manager should be aware of the solutions to diverse threats posed by the long-running cooperative agreements, such as the entry of new participants in the fishery. In this scenario, solutions such as *transferable membership*, a *waiting period* \cite{47} and the *fair sharing rule* \cite{48} have been proposed. Empirical analysis of the new member problem in the fishery of the North Atlantic bluefin tuna shows that the threat of a new member entry is not sufficient to break down the existing cooperative agreement, given the low level of the stock, which makes non-cooperation a low-payoff strategy. The authors maintain that, if the cooperative strategy calls for an initial harvest moratorium, then the threat becomes progressively more relevant. At this stage, from simulations, the authors conclude that *transferable membership* and the *fair sharing rule* options present the best solution for the new member entry threat \cite{49}.

The manager should be also aware of the means to ensure the stability of a cooperative arrangement over time, which is known as *time consistency*. In order to achieve this aim, economists have developed the concept of the *agreeable transfer payments programme*. Such a programme provides enough information to support the decisions. This will be especially useful when assessing the negotiating costs of implementing a cooperative agreement (McKelvey and Golubstov 2002, cited in \cite{30} p15).

Managers should also hold information on the costs involved in progressing from cooperation in research to active cooperative management. In this stage, managers have to take decisions on setting and allocation of TACs, effort limits and fishing capacity management. These secondary levels of cooperation involve additional costs in the areas of:

1. Decision-making: for example, stakeholder consultations and preparation of fisheries management plan at national levels.
2. Management plan implementation and enforcement: for example, putting in place regulations, seeking cooperation from industry and fishers, and monitoring, control and surveillance.
3. Evaluation of management regimes: for example, assessment of whether objectives have been reached, investigating reasons for not meeting targets, and developing measures for improvements.

\footnote{With transferable membership, a new member is allowed to enter the exploitation if another member relinquishes its share. With respect to the waiting period, if a new member was allowed to enter the exploitation, it would have to wait a certain period until enjoying benefits from the fishery.}

\footnote{The last solution depends on a fair sharing rule based on the contribution of each partner to the coalition.}
2.5.5 Knowledge of technological factors

Since effective MCS actions are required to control activities of foreign fleets in national EEZs, MCS systems require accurate technological means to back up their activities. Information about the technical capabilities (for example, patrol boats, skilled personnel, satellite tracking devices) required to control the activities of fishing vessels should be taken in account when negotiating the joint exploitation of fish stocks. This information will allow the hosting states to know in advance the requirements needed to accurately control the activities of the various fleets that will operate in its waters. This would help the coastal state to assess if the investment in MCS improvement would be compensated by the gains derived from joint exploitation. In some cases, it can be worthwhile for the coastal state to receive advice from the partners with better technology and expertise in MCS issues on how to improve its MCS system. This could be part of the agreement. MCS capabilities also play an important role in managing shared stocks in coastal areas because air surveillance or satellite devices help to prevent illegal activities. These capabilities allow MCS systems to monitor fleet activities and prevent encroachment of industrial fleets into reserved areas, such as areas reserved for artisanal fisheries and sanctuaries.

The issue of the technological creep of fishing fleets is another factor that requires attention and research. Research should assess how fleets’ fishing power, efficiency and capacity evolve over time. It is also important to assess how over-capacity threatens both the already exploited resources and potential ones. Technological factors should also be taken into account when allocating resources among participants. The rationale is that the more developed countries may hold a more advanced technology, which allows them to harvest at a higher rate. This may be to the detriment of the others contractors. Moreover, the analysis of the technological factor is essential to accurately design arrangements between a developed DWFN and a coastal state. The underlying rationale is that the less developed nation may not obtain benefits from the exploitation of certain resources without the intervention of developed fishing nations. This happens due to the fact that the former does not hold the technology, expertise and funds to carry out prospective research and exploitation. Furthermore, meaningful benefits may come through transfer payments, training of local fishermen, transference of technology, etc. Finally, the exchange of relevant information regarding technological capabilities is a key factor in the development of effective agreements, especially in the process of data gathering, assessment, and MCS.

2.5.6 Knowledge of social factors

In order to take effective decisions in the management of shared stocks, it is essential for managers to hold knowledge about the socio-economic importance that concerned countries place on shared resources. This knowledge will help in the allocation process since countries usually place a different valuation on the resources; this fact is one of the main tasks to be faced within this process. A typical case is the shared exploitation of cod by Norway and Russia, which is related not only to the economic value of cod exports for Norway, but also to the social importance that traditional consumption of cod has for the diet of Norwegians. Another factor that can determine the value
a country places on a resource is the dependence that coastal communities may have on a given fishery. This dependence can take the form of subsistence fisheries, in which fish consumption is essential to maintain a community’s way of life. However, the social knowledge base required to conduct accurate management relates not only to the social value a country places on fish, but also to knowledge regarding fishers behaviour in a scenario of shared exploitation of the resource. This knowledge is essential to understand the mechanism that leads to either compliant or non-compliant behaviour and helps to establish accurate international arrangements. Knowing in advance how the various parties would behave may help to establish the basis for an accurate agreement: for instance, to devise agreements that have an accurate duration, well-specified rights, transparent rules of access to the resource and effective threats to deter cheating. The setting of accurate agreements will enhance legitimacy, it will reduce non-compliant behaviour, and this, in turn, will reduce enforcement costs.

2.6 INSTITUTIONAL SUPPORT FOR KNOWLEDGE PRODUCTION, ADVICE, COMMUNICATION AND DECISION-MAKING

Scott considers institutions as cognitive, normative and regulative structures and activities that provide stability and meaning to social behaviour [50]. Regarding knowledge, Scott states that institutions validate what is true knowledge: this being rules, norms and knowledge, which are the essential elements of institutions. Because of the high prestige of science, scientific knowledge is regarded as a strong objective support of management decisions. As Gray [51] suggests, policy makers need for scientific support to deliver regulations and take decisions is widely accepted within the fisheries management system. The role of scientific knowledge produced by research institutions is fundamental in the management of shared stocks and, in many cases, serves as a basis for active management—the process which is here termed the primary level of cooperation.

In the management of shared stocks, scientific knowledge is created within a wide range of scientific institutions, such as national research institutes, universities, and advisory bodies. In the case of the Northwest Atlantic Fisheries Organisation (NAFO), for example, knowledge is produced by contracting parties and then brought to NAFO for assessment and management. In the case of the EU, the chain of knowledge starts at local research institutes that are, in most cases, state institutions which produce knowledge by mandate from national administrations. This nationally produced knowledge is then collated by the International Council for the Exploration of the Sea (ICES) which conducts assessment of the national findings. This assessment is, in turn, revisited by the EU’s Advisory Committee for Fisheries Management (ACFM) who communicate their conclusions to the European Commission. At this point, ACFM’s conclusions are also examined by the Scientific, Technical and Economic Committee for Fisheries (STECF). Finally, the Commission reports to the European Council of Ministers which makes the final decisions on, for instance, annual TACs for commercial species. ICES also acts as a scientific advisory body to contractors outside the EU, such as the Norwegian-Russian Commission. However, in this case national research institutions carry out research, whether in cooperation or independently, that performs as an input to ICES’ advisory
role. ICES also acts as an advisory body to North East Atlantic Fisheries Commission (NEAFC).

Finally, with regard to the question of institutional support, it is important to highlight the role of international agencies, such as the UN, which implement legal instruments such as the Law of the Seas Convention, which states the need to cooperate in research as well as setting the rules for management of shared stocks. The FAO also has an important role in the coordination and sponsorship of research, dissemination of knowledge and guidance in shared stocks management. Important initiatives include the October 2002 Norway–FAO Expert Consultation on the management of shared stocks, which gave participants from both developed and developing countries the opportunity to deliver knowledge and exchange experiences about the management of diverse shared stocks all over the world.

2.7 CONTROL, ENFORCEMENT AND COMPLIANCE

According to Gulland [1], without accurate implementation and enforcement the best cooperative arrangements can be useless. If the arrangements that support cooperation are transparent and enforceable a sustainable exploitation of the resource can be expected. However, unclear or bad agreements lead either to unilateral exploitation or to heavy exploitation of the resources by all interested parties. An agreement must contain a clear set of rules for allocation, cooperation in research and transparent rules for exploitation and a strong means of control and enforcement. This can be problematic in cases in which a developing coastal state is to monitor the activities of well-developed DWFNs. Fishing in the high seas is also a challenge for enforcement agencies. In spite of this, innovative approaches and goodwill can be key factors for success for coastal states aiming to strengthen their MCS systems. Accurate agreements not only ensure involved coastal states comply with the rules but also allow them to face threats collectively. The case of the Pacific Island nations is a good example of cooperative management as a basis for effective MCS. Even though these developing nations have difficulty affording sophisticated aircraft and patrol vessels, they have managed to establish a regional register of foreign fishing vessels. One vessel found in violation of its access terms and conditions in the waters of one state faces the prohibition to enter the EEZs of the rest of the contracting nations.

Focusing on transboundary stocks, Munro et al. [11] draw four scenarios for implementation and enforcement. In the first scenario, which is the simplest, two coastal states share a given resource. They have settled all boundary delimitations and there are no third countries fishing the resource. In this case, effective implementation and enforcement of fisheries management regimes requires the following measures:

(a) Maintenance of a vessel register.
(b) Use of a system to monitor fishing activities, which should include fishing authorisation, quotas, and records, perhaps in logbooks, of fishing locations and trip durations.
(c) Port inspection of vessels, and both onboard and post-unloading inspection of the catch.
In the second scenario, two participants dispute an area. This is the case of the Norwegian-Russian joint management in the Barents Sea [27]. There is dispute for an area of 155,000 square kilometres. The Russian and Norwegian authorities established a ‘Grey Zone’. For this area the two countries agree upon harvest quotas and each coastal state regulates its own fleets. The FAO report suggests that, besides the measures recommended in the first scenario, the following measures are required for the Grey Zone:

(a) Special reporting requirements for fishing vessels when they operate in the Grey area, including the provision of simultaneous reports for both coastal states.
(b) Reciprocal monitoring and surveillance schemes for each coastal state with respect to vessels flying the flag of one of the parties and operating in the Grey area, or primarily flag state responsibility for monitoring and surveillance.

The third scenario is similar to the first one, the difference is that the coastal states, allow third countries access to shared stock fishery resources. This third state could be a DWFN, and is the case in the region of Svalbard. In this case, the following measures, designed to prevent free riding, are required:

(a) Activities by a third country’s fishing vessels should be controlled and subjected to surveillance by that coastal state.
(b) Third-party vessels should be subject to at least the same terms and conditions of license (boarding, inspection and enforcement requirements) when they are operating in either or both EEZs, as the terms and conditions imposed on national vessels from the coastal states.
(c) The flag state legislation should include control measures for its vessels fishing in the EEZ of another state.

In the fourth scenario, the situation is identical to scenario 1 except that the coastal states grant reciprocal EEZs access to each others’ fleets. The measures required are similar to those describes for the three scenarios outlined above, plus:

(a) Catch and area of operation reporting and notification of entry and exit from the EEZs, both of which would serve to reinforce coastal state management measures.
(b) Collaboration between the coastal states in order to sensitise fishers and incentivise them to abide by the terms and conditions of their licenses.
(c) A means that can be invoked by one of the coastal states in the event that its vessels commit an offence in the adjacent EEZ.
(d) Observer programmes for scientific and enforcement purposes.
(e) Education and awareness-promotion of regulations for fishermen.

The issue of compliance is an important source of concern among policy makers and managers. Understanding the mechanisms that lead to compliant or non-compliant behaviour is a matter of considering aspects related not only to the perception that it depends on the expected gains from illegal practices, but also that those aspects are embedded in a social context in which, inter alia, enforcement, participation and moral factors play a
major role. Compliance depends on legitimacy: if fishermen hold the perception that the decision-making process is taken under uncertainty due to insufficient information, they will get the impression that the outcome and the process is unfair. This will have the effect of reducing legitimacy and will bring about non-compliant behaviour, which in turn increases management costs [52].

It seems that compliance strongly depends on the perception the involved parties have of their right to fish. Weak property rights, particularly those lacking the accurate duration, can incentivise the actors to overfish quickly due to the fear of losing both their rights to fish and their intrinsic benefits. Queirolo et al. [16] argues that once a DWFF acquires the right to fish during an indefinite tenure, the coastal state would no longer have monitoring problems and the problem of asymmetrically held information by the DWFN would disappear. According to this argument, long and secure rights-tenure would bring about compliance and, therefore, ease enforceability. However, it is risky to assume that having such a tenure of the right to fish would preclude the DWFN from attempting to harvest more than their quotas allow, to fish undersized species, to discard, or to fish in times and places where harvesting is prohibited. Thébaud [29] considers non-compliant behaviour as one reason for the failure to achieve long-term cooperative solutions. He suggests that a solution to this could be to establish a ‘system of credible threats’ as one of the features of the arrangement: for example, the possibility to revert to competition in case of defection.

2.8 CONCLUSIONS

Management of shared stocks becomes a very complex task when attempting to achieve long-term fishery system sustainability. Even though sustainability is a common objective among interested coastal states, there exist particular and often conflicting national management objectives. Therefore, asymmetry of objectives, knowledge, and technical capabilities is commonly encountered in a joint exploitation scenario. In many cases, the asymmetry problem and the inconsistency of regulations and agreements contribute to the creation of controversy and conflict.

Cooperative fisheries arrangements are acknowledged as the best means to manage shared stocks through time. Conversely, as experiences all over the world suggest, non-cooperative management arrangements lead to over-exploitation. It is necessary to highlight that cooperative arrangements have to be binding to ensure their stability over time, in case the underlying conditions change. A cooperative process often begins with cooperation in research and data gathering. However, the potential benefits from cooperation not only depend on the nature of information (symmetry/asymmetry) held and shared by participants, but also on the quality of that information. In this context, knowledge production must consider not only scientific information related to biology, ecology, and socio-economics, but also users’ knowledge.

Shared stocks management schemes require biological knowledge related to the distribution, migration patterns and stock structure, among other indicators. This kind of information influences economic variables, such as costs of harvesting and values of catches. Economics provides a powerful tool to resolve shared stocks conflict by integrating
game theory using standard bio-economic models of fisheries. The two most important economic factors that determine the nature (optimal/suboptimal) of the game solution (or the management problem solution) are harvesting costs, and the discount rate. However, this economic information must be supplemented by other social knowledge such as consumer preferences or, more generally, the socio-economic importance that coastal zones place on the shared stocks. This social knowledge must be interpreted in a more extended form, comprising both the social value a country places on fish, and knowledge of fishers’ harvesting behaviour. The management of shared stocks also requires ecological knowledge for the sustainability of fisheries, such as the impact of fishing gear and activities on the habitat. Finally, technological factors, which are closely related to ecological impacts of fishing, are part of the required knowledge basis for shared stocks management. In particular, it is important to analyse how fishing power, efficiency and capacity impact on resources.

Allocation being the most difficult process when managing cooperative agreements, social, technological and other bio-economic factors should also be considered as an important input in the allocation process. Managers have to hold information regarding consumer demand, needs of the fishing industry, dependency on fishing, structural inflexibility of fishing families and communities, fishers’ behaviour, value of the fishery, state of fishing technology and MCS capabilities. These considerations could give both quantitative and qualitative arguments which would help in the deliberation over resource allocation. Finally, even good cooperative agreements will become inefficient without an appropriate control and enforcement system that can assure a sustainable exploitation of resources by the countries concerned.

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3.1 INTRODUCTION

Since the 1980s there has been a gradual movement towards market-based fisheries management systems and a parallel decline of conventional state-based command and control regimes. Market-based systems are specific right-based formulations, which rely on market forces to internalise the externalities found in fisheries resources. Although other solutions are also possible (for example, individual effort or input quotas, individual territorial quotas or even group quotas) individual transferable quota (ITQ) programmes have monopolised the theoretical, empirical, and also the political, debate surrounding market-based fisheries management.

The attractiveness of ITQs is the result of two related issues. First, since ITQ systems are based on a long-standing traditional instrument—total allowable catches (TACs)—their adoption does not, apparently, involve a total rupture with the status quo. This implies the possibility of taking advantage of much of the biological research-based knowledge surrounding overall quota setting. Second, given that ITQs are TAC-based, they are able to simultaneously address both biological and economic goals. Additionally, with the initial allocation of individual quotas and the introduction of transferability rules, certain distributional goals may be pursued. Last, but not least, ITQs sail in favour of the wind, in a world economic policy framework highly inspired by neo-liberal philosophy.

ITQs fit very well into the neo-liberal economic thought sown by Adam Smith in his The Wealth of the Nations [1], when he wrote: “the invisible hand of the marketplace guides selfish agents into promoting general economic well-being.” Thus, individuals are best left to their own devices, “without the heavy hand of government guiding their actions”. That philosophy was propped up by Debreu’s [2] formalisation of the invisible hand argument, known as the first fundamental theorem of welfare economics, which stands that, under certain conditions (price taking behaviour, perfect information, rivalry and excludability), the market mechanism is able to generate an efficient allocation. Moreover, the second fundamental theorem of welfare economics shows that any efficient allocation can be
achieved by a market mechanism through an appropriately redistribution of agents’ initial endowments. Thus, by simply transferring endowments, governments would achieve any distributional goals, leaving the market to attain efficiency. Normatively, the welfare theorems’ provide the following advice: use the competitive mechanism, use the free market; do not use prices to attain distributive goals, use the endowments.

Although, broadly speaking, empirical evidence suggests that ITQs have been successful in the economic rationalisation of fisheries, this has not necessarily been a win-win solution. Efficiency and distributional trade-offs are difficult, if not impossible, to avoid. Besides, the approach taken to establish exclusive user rights, and, therefore, the specific consequences, varied significantly from place to place and even from fishery to fishery. Thus, evidence suggests that, where ITQs are introduced, the institutional framework, the history, the traditions, the stakeholders’ attitudes, the biological features of the species, the situation of the stocks, and above all, the policy objectives, determine not only the choice among different market-based individual quota approximations, but also the results of the system as a whole.

There seems to be a link with the objectives managers want to achieve, the hierarchy among these objectives and the accepted knowledge base in a management system. To put this another way, the choice between objectives determines the inclusion and validation of certain types of knowledge. Furthermore, the hierarchy of objectives establishes the hierarchy of the types of knowledge. For example, it is broadly admitted that the first objective of an overall quota system is stock conservation. Thus, it is easy to understand that the primary science is usually biology. When also including socio-economic goals (as is the case of ITQs), social sciences are expected to be progressively included in the knowledge base for fisheries management.

The main question raised in this chapter is: What is the knowledge base for a right-based fisheries management system (RBFMS)? First we will turn to the main features of an ITQ system. Special attention will be paid to the knowledge derived from the empirical evidence of various operational frameworks.

3.2 MAIN FEATURES OF AN ITQ SYSTEM

ITQs face externalities arising from non-excludability by introducing a system of exclusive use rights\(^1\) above individual shares of an overall quota (TAC), established by the institution holding the collective choice rights. This TAC is divided into individual allocations (IQs), which give holders the right to catch a specific quantity of fish, usually a percentage of the TAC\(^2\), from a given stock within a given time period. The security on individual catches ends with the incentives to fish as quickly as possible before the TAC is filled (i.e., the race for fish). Operators with IQs then have the flexibility to increase

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\(^1\)Exclusive use rights are considered pseudo property rights. They do not confer ownership of the fish stock itself; Rights are limited to the resource flow.

\(^2\)This way the risk associated with variations in the fish stocks is supported by the industry.
their profits by lowering their costs to find the most efficient way to harvest fish and by increasing their revenues, spreading their effort optimally across the entire season and avoiding offer gluts by selling their products at those times and in those markets where prices are higher.

Allowing the individual shares to be transferable (ITQs) means the creation of a quota market where quota transfers take place. With no limits on transferability, individual operators are free to optimise the scale of their operations by buying or selling quota. The price mechanism governing the quota market (i.e., the invisible hand) will lead to an efficient redistribution of quotas, whatever the initial allocation is. In the long-term quotas, would be consolidated in the hands of the most efficient operators. Divisibility of quotas would promote short-term rationalisation.

The longer the time horizon of the quotas, the more probable it becomes that the optimal investment strategies (affecting both fleet capacity and resource conservation) would be adopted. With a quota allocation that is valid for a sufficiently long time (at least the lifetime of a fishing vessel), the quota holder can make rational predictions about future catches and determine the fishing capacity accordingly. Since ITQs are a market-based mechanism, it seems rational to assume that agents have rational expectations and thus, that ITQ prices convey information on market expectations over a range of variables that co-determine profitability. Long-term tenure of quotas is also important for taking on resource conservation strategies. Long-term quotas reach higher market values than short-term ones, especially if the resource is properly managed, which strengthens the dependence between quota values and the expected future productivity of the stock.

As described above, ITQs seem to generate incentives to provoke efficiency gains in the fisheries. But ITQs may also induce negative procedures, such as discarding. Once the race for fish is finished, fishing operators may pursue to maximise the net value from their quota, filling it with the best quality of fish and discarding the lesser quality one. This waste of fish not only would diminish the overall net revenue obtainable from the fishery, but also, as discards are seldom reported, may lead to the decline in the quality of data to be used in the assessments.

ITQs also may generate important changes in welfare distribution, depending on the initial allocation and the transferability rules adopted. That is why, among the fundamental issues to be addressed when designing an ITQ system (i.e., the management units, TAC determination, the initial allocation, transferability rules, monitoring of enforcement

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3 On board labour and security conditions may also improve as a result of introducing ITQs. With no need to race for the catches, fishermen are no more pushed to operate in bad weather and dangerous conditions.

4 With individual non-transferable quotas, despite the positive effects the elimination of the Olympic fishing would imply, fishermen could not optimise their operations by buying or selling quota. Therefore, less efficient fishermen will be reluctant to leave the fishery since they cannot sell their quotas. Thus, limiting transferability in individual quota programmes has a cost in terms of potential efficiency gains in the fisheries.
design, the potential need of other regulation, resource rentals and cost recovery\(^5\) the criteria for the initial allocation and the transferability and duration rules are especially controversial. The initial allocation establishes the initial participants in the fishery, while the rules for transferability and the duration of the ownership rights determine the future participants (i.e., who gains from the fishery in the long run). Hence, the distributional short and long-term implications and also the potential efficiency gains are to a large degree determined by the decisions taken at this stage.

Albeit that there are many initial allocation options, usually historical catches and the current level of investment (i.e., the size of the vessels) form the basis for quota allocations, almost as a rule directed to the vessel owners and given away free. Arguments against transferability focus on the problem that certain individuals are obtaining wealth from a public resource; while opinion in favour of free transferability is linked with maximum potential efficiency gains. Limiting transferability to a greater or lesser degree is the general practice in real-life ITQ examples. Limits are established to prevent distributional effects and also to avoid excessive concentration of shares.

Depending on the overall objectives the specific ITQ management system would like to achieve (i.e., resource conservation goals, economic efficiency goals, certain distributive goals, etc.), different criteria and rules are to be adopted to fit the biological conditions of the resources and the broader socio-economic, political and institutional frameworks. The initial allocation and transferability rules will determine the final structure and characteristics of the fishing sector, which should be previously agreed based on social welfare preferences. The fisheries policy objectives and the hierarchical structuring among these objectives (trade offs cannot be avoided) constitute the main determinants and restrictions of the ITQ system configuration.

For example, the introduction of ITQs in New Zealand fisheries was a part of a process of trying to rebuild fish stocks and making fisheries as economically efficient as possible. Moreover, ITQs were also a piece of a general economic reform during the 1980s; carried out to get a more competitive and open to trade economy, through lowering tariff barriers and dismantling subsidies [3]. The history of New Zealand’s fishing industry and the fleet’s structure are also remarkable. For one side, the grounding of the fishing industry in local communities is rather limited, and, for another, the fishing sector is mainly constituted by large-scale corporations of a relatively young deepwater sector [4, 5].

\(^5\) Since ITQs incentivise fisheries rationalisation processes, the question is raised of whether some (or all) of the rents should be extracted for the public coffers, either to pay for the costs of managing the system (cost recovery) or to insure that the gains from a resource that belongs to the entire nation are shared by all citizens (resource rentals). Although this is strictly a matter of public policy, it is important to put stress on the fact that rent extraction and cost recovery are separate issues. One could argue that cost recovery should be included in all management programmes, whether they are rights-based or not. However, one important problem with cost recovery programmes is the difficulty of estimating what part that the government spends on fisheries is attributable to management and how it can be allocated to the different species under management. If this cannot be done, it may be wise to impose a landing tax as a percentage of the ex-vessel price [39]. For example, in Alaska, 3% of ex-vessel value of IFQ harvests are collected to cover management and enforcement costs. In Australia, national policy requires 90% recovery of management costs attributable to managing all the fisheries, irrespective of whether they are managed by ITQs, while in New Zealand there is a full management cost recovery from industry.
this framework, it is not accidental that ITQs in New Zealand are the most firm property rights one finds in ocean fisheries anywhere. They are explicitly perpetual, they can be transferred almost without restrictions (there is a limit on how much quota can be held by any one firm, and foreign interests cannot own ITQs), and they can be mortgaged.

Norway, with a considerable history of individual fishing rights, also offers an interesting picture. Norwegian fisheries regulations are focused on four main objectives: increasing the profitability of the fisheries sector; protection of resources; securing employment opportunities in coastal communities; and maintenance of coastal settlements [6]. The two latter objectives are closely related, since employment opportunities are required for people to continue living in the fishing-dependent communities along the coast. There is also an obvious trade-off between efficiency and social-regional considerations. Following efficiency goals, the rights system should be designed to incentivise a cost-minimising modern factory-vessel fleet, but this could threaten the fishing communities. If one of the accepted roles of fishing is to provide employment and maintain communities in areas where there are few other opportunities, free transferability seems not to be the solution. In fact, according to the 2002 plan to introduce individual vessels quotas for smaller boats, short-term transfers will be subject to the authorisation by the Ministry of fisheries, and long-term transfers of quotas will be not permitted for vessels less than 15 ms [3].

Assuming good enough knowledge regarding the size and growth of the fish population, the behaviour of fishermen and other stakeholders and a robust enforcement system, ITQs provide strong incentives to maximise rents in the fishery and also to reach certain conservation targets establishing an appropriate bio-economic TAC. Nevertheless, as the case of Norway demonstrates, trade-offs cannot be avoided. Moreover, trade-offs should be identified, as far as possible, prior to decision-making. Generally, the strongest form of property rights with the fewest transferability constraints will be able to achieve the maximum economic efficiency regardless of the initial allocation (as the second welfare theorem shows). Restrictions on transferability to achieve certain social, demographic or distributive objectives will bind Smith’s invisible hand; the efficiency goal will be sacrificed; the fishery will attain a lower efficiency level. Thus, there is a trade-off between potential efficiency gains and restrictions on transferability; a specific formulation of the typical efficiency-equity trade-off in economic policy.

From reviewing operational quota-based market systems, it is apparent that ITQs really do result in efficiency gains, lowering the private and social costs of the race for fish. Different indicators, for instance the level of employment in the fishing sector, the number of the vessels, the fleet’s capacity, the volume and value of the catches, and also the evolution of the individual quota prices are expected to suffer potential variations depending on the real framework and initial status of the fisheries. We would expect less capital invested in fishing boats, fewer fishermen and increasing quota prices as a result.

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We are referring to the TAC that maximises the net social benefits from the fishery. But, even if the TAC determination only pursues biological targets, ITQs can address not only the biological aspects of management; ITQs are also able to internalise the negative externalities derived from non-excludability. The establishment of exclusive, secure and transferable rights would be enough to guide the market forces to acceptable (although not maximum) efficiency levels.
of increasing efficiency. The evolution of catches is more related to the TAC setting, but taking into account the increasing influence of the fishing sector in the determination process, rising catches could be (although not necessarily) interpreted as a warning sign of proper management.

The British Columbia halibut individual vessel quota programme (1991) provides a good example illustrating the significant improvements in economic performance coming both from higher revenues (as a result of better meeting the needs of the market) and from lower fishing costs [3, 7]. The increase of the fishing season was spectacular (from 6 days to 214 days), and raised the fresh-marketed share from 50% to 90%. As catches were absorbed by the fresh fish market, rather than being frozen and turned into an inferior product, the value of the catch increased considerably. Once transferability was permitted, the number of vessels participating in the fishery started to fall significantly and so did the demand for labour in the harvesting sector. Employment of shore workers declined slightly, because there has been a shift from frozen product to fresh halibut, which requires less shore-based employment.

A second example is to be found in the Dutch flatfish fishery. On the introduction of transferability, the total number of Dutch flatfish ITQ holders decreased significantly by nearly 30%. This was due variously to the selling of ITQs, decommissioning, operators ceasing their enterprise for other reasons, or switching to an alternative gear-type. Now, nearly all large shareholders own more than one vessel. There was a concentration tendency in the period 1988–1993 (3% of quota holders owned 17.4% of the total quota in 1994); although after this time the trend was less clear-cut [8].

Meanwhile, in the USA’s Surf Clams and Ocean Quahogs fisheries, in the little over 10 years since the ITQ programme took effect (1990), similar effects indicating efficiency improvements were perceived. The fleet reduced by a half, both in numbers and in tonnage (in 1999 there were 41 vessels in the surf clam fishery, compared with 141 vessels ten years previously). The hours fished per vessel per year more than doubled (from 154 hours per vessel to 380 hours per vessel). With fewer vessels, despite the fact that the remaining vessels are now used full time, a 30% reduction in the number of jobs occurred [9].

Finally, a quick review of ITQs in Iceland indicates that the system has in fact increased the fleet’s economic efficiency [10]. The number of vessels has severely decreased since the introduction of quotas, although the fleet capacity (in terms of GRT and HP) has been gradually increasing despite the ITQ system. The inshore fleet (defined as vessels from 12 to 200 GRT) has seen a great reduction (27% of GRT), while the trawler segment has increased (33% of GRT). The capacity increases can be explained by the growth in the factory trawler fleet, which, as a result of having on-board processing facilities, loses any comparability to other vessels. In terms of export value of fish products and profits made by leading fisheries companies, there is little doubt that the ITQ management has been success. Icelandic fishing companies are expanding into international waters and demonstrating competitiveness in terms of technology and know-how.

Let us now focus on the distributional concerns that initial allocation and transferability rules may undertake to address. ITQs seem to have increased the bargaining position of quota owners (often the vessel owners) at both micro (inside individual boats) and macro
levels (like an overall pressure group). One reason for this is that the cost minimising rationalisation means crew savings and, therefore, more power for the owner. A second is that the eradication of pressure to “race to fish” may replace owner’s incentives for the traditional income sharing remuneration system with self-interested modifications. As a result, incentives for skilled crew to stay on may be reduced, and, paradoxically, although jobs may be fewer as owners economise, the quality of crew willing to work may be reduced [10].

The conflicts between vessel owners and crewmen have been at the centre of the debate over Icelandic ITQs, since the cost of leasing quotas was deducted from the catch value as an operating cost before calculating the share of labour (i.e., quota leasing became a common cost), which obviously caused a decline of the crew’s share of the catch. Moreover, some operators who own quota shares rented out their own quotas and based their operations on quotas leased from others, in order to legitimise claims to deduct leasing costs. This phenomenon, which has also occurred in the USA Surf Clams and Ocean Quahogs fishery causing much controversy, provided the background to the Icelandic crewmen’s strike in January 1994, as well as repeated strikes in 1995 and 1998. After many rounds of negotiation, a new institutional framework was set up in March 1998 to control prices, resolve disputes and control leasing transactions [11].

Since ITQs are an economically oriented management instrument—specially designed to incentivise individual cost-minimising strategies and to provide individual-profit maximising allocations of fishing resources—changes to the fleet structure, quota concentration and absentee ownership are natural and ex ante expected phenomenon of the rationalisation process that may or may not be socially desirable, depending on the institutional, political and socio-cultural framework. Concentration has a positive side linked with making good use of economies of scale in the fishing industry. Its negative effects are likely to be associated with power in the labour market in frameworks characterised by lack of employment alternatives or monopolistic local power in the consumer markets [3].

In Iceland, since the introduction of ITQs, the inshore fleet (defined as vessels from 12–200 GRT) has suffered a 27% reduction in tonnage, while that of the trawler category has increased 33%. There has also been a substantial concentration of quota shares within the larger, vertically integrated companies. Although, on the one hand, the losing of the weight of inshore fleet and related structural changes can be interpreted as clear evidence of the rationalisation of fisheries activity; on the other, the phenomenon could cause additional social breakdowns, which are more substantial than the economically accountable income losses of the small-scale artisanal fishermen. Concentration of quota ownership and larger individual enterprises could mean more profitability for the fishery sector, and simultaneously pose a threat to smaller coastal communities. Responding to the growing concentration of quota ownership, in 1998 Icelandic government agreed an upper limit to the TAC shares that can be held by a single owner.

The US Surf Clams and Ocean Quahogs fisheries, in which quotas are not tied to specific vessels and there is no upper limit, have provided a context for interesting structural changes. Some vertically integrated processors have divested themselves of vessels, instead holding quota shares to ensure supplies and letting independent operators do the fishing. Although, apparently, the number of quota holders has not changed
much, the number of vessel owners has fallen from 60 to 21, which constitutes a clear
evidence of absentee ownership of quotas. Additionally, there is substantial concentration
of ownership: 10 quota owners (20% of the total) own 70–80% of the quota shares.
However, the eastern seaboard of the United States is not exactly characterised by lack
of employment alternatives and clam products are sold in fierce competition with other
products, so even a single producer of clams would not have much market power in the
consumer market.

Long-standing fisheries-dependent communities, in which most fishers are small-scale,
are often afraid of losing rights of access and also of geographic shifts in fish landings,
the location of processing firms, and changes in ancillary industries (such as welding
and ice-making) that can be generated and/or accelerated by ITQs. Interestingly, some
communities have obtained (for example, the Chatham Islands, New Zealand) or are trying
to obtain (Gulf of Alaska) ITQs in the name of the community rather than individuals,
in order to gain more control over the transfer and distribution of quota and, hence, over
opportunities for jobs and income.

In Iceland, small communities have on average lost a much larger share of their quotas
comparing with bigger ones. Land-based frozen fish production is being substituted
by processing at sea and an increase in exports of fresh products. Consequently, the
local freezing plants are no longer a guarantee for direct and indirect employment. Of
course, not only ITQs, but also changes in technology and markets, have contributed to
this crisis. ITQs have probably accelerated the process, as well as generating a strong
demoralising effect on the inhabitants of small fishing communities resulting from losing
their right to catch fish, which in most cases is not even partially mitigated by the
compensation of selling quotas when exiting the fishery. Contrary to boat owners, fish
workers, crewmembers and other community residents hold no valuable rights, so they
are excluded from even that recompense mechanism.

According to conventional neo-classical economists, ITQs are the most efficient mea-
sure for dealing with the fisheries problem. However, distributional issues, high trans-
action costs, additional problems related to discarding, as well as potential enforcement
difficulties, may outweigh the positive effects of this management instrument. If expected
efficiency gains are sufficiently high to counteract the management costs, ITQs may be
the solution to the fisheries problem. But, even if the cost-benefit proof is successfully
demonstrated, empirical evidence shows that, sometimes, political problems and lack of
knowledge (i.e., information problems) may be difficult barriers to overcome. Special
attention should be paid to the knowledge base necessary for successful ITQ management.

3.3 THE KNOWLEDGE BASE FOR ITQS

The over-riding generic goal of any fisheries management instrument is the long-term
sustainable use of the fisheries, which requires optimising the benefits derived from
the resources. But this is a general and vague objective that needs to be defined by
collective choice rights owners’. More specifically, the goals in fisheries management can
be divided into four general subsets: biological goals, specially linked to conservation
of the resources; economic goals, related to economic efficiency; social goals, highly
connected to the distribution of welfare; and ecological goals, aiming to minimise the impacts of fishing on the physical environment. Examples of combinations of the above mentioned goals may be to maintain the target species at the levels necessary to guarantee their productivity, to maximise the net incomes of the participating fishers or to maximise employment opportunities for dependent communities.

ITQ systems are particularly related to biological and also socio-economic goals. Setting the goals, identifying different trade-offs (both fundamental biological-economic, economic-social trade-offs; and short-term vs. long-term trade-offs), and deciding the importance or hierarchy among them is an essential, and frequently controversial, task. This is because there are few examples of win-win outcomes among different stakeholders who try to influence the decisions, which are usually taken by politicians.

Assuming that the TAC is exogenously determined using the best available knowledge in order to attain certain conservation goals, the individual quota system primarily stands to obtain the maximum economic benefit from the overall quota. Thus, the TAC is mainly related to conservational goals, whereas the division of overall quota in individual tradable shares is related to economic efficiency. Furthermore, in the described scenario, economic efficiency seems not to conflict with any conservationist goal. To a certain degree, the objective to distribute welfare may be catalogued as an additional indirect objective, which is realised through the initial allocation of quota, restrictions in transferability, the setting of limitations on eligibility and the success of the enforcement system.

The knowledge used in a management system seems to be related to the objectives the managers want to achieve and also to the hierarchy among these objectives. Thus, the hierarchy in objectives determines the hierarchy of the type of knowledge. The choice between objectives determines the choice of certain types of knowledge and, consequently, the inclusion/exclusion of those types of knowledge. That the setting of objectives also requires knowledge is clear: ranking objectives and choosing specific knowledge types is a normative process that reflects certain types of leading preferences that are linked to the worldview of the decision makers. In the next sections the knowledge related to each of the above-mentioned objectives will be described.

3.3.1 The knowledge base for TAC setting: biological objectives

3.3.1.1 Developments in research based knowledge for TAC setting: rising complexities and facing uncertainties

In an ITQ system, it is essential to set the TAC, which is typically established under biological science and criteria. In order to be able to do so, and depending on the underlying model to be adopted (production model, size and age-based model, or stock recruitment model, or any of their multispecies extensions) one needs accurate biological data (biomass estimation, annual catches, growth rate, natural mortality rate, number of fish caught per age class, or even separate estimates of stock and recruitment over a number of years, etc.) and knowledge about the population dynamics. This knowledge is however, not always available.

This was, for instance, the case in the South-East Fishery of Australia, where the initial stock assessment and TAC-setting process was ad hoc and rushed. For most species,
TACs were not based on formal estimates of relative abundance, but were instead based on average catch levels over the period 1986–1991 [12]. In the initial stage of the introduction of ITQs in New Zealand in 1985, “carving up the New Zealand EEZ into ten areas with the concomitant distribution of stocks and sub-stocks was done based on rather rudimentary knowledge of species distribution. Even more insecurely based was the preliminary fixing of TACs where biologists of the Ministry of Agriculture and Fisheries (MAF) had long-term time series for only a small number of the stocks involved. For the others, it was a question of educated ‘guesstimates’ as admitted by one of the early participants. However, the whole procedure was designed so that adjustments could be made en route as more information became available” [13].

When analysing the research-based knowledge produced by formal science institutions in the TAC setting process, what we see is the development of a knowledge base from a rather simple model towards an increasingly complex one. This progression towards complexity is being developed under pressure, because a variety of ecosystems and fish stocks are on the verge of collapse. Fisheries scientists and managers recognise the complexity of these situations and try to tackle it by incorporating more knowledge of a variety of factors into the scientific advice. Much more than traditional stock assessment is required today, accepting incomplete knowledge and uncertainties in the knowledge base for fisheries management. Paradoxically, the growth of fisheries knowledge has brought with it an increase in the uncertainties surrounding fisheries management, some of which are not easy to face, even with the complex methodologies developed in the last decades.

Following Charles [14], uncertainties in fisheries arise in three principal forms: (1) random fluctuations; (2) uncertainty in parameter estimates and states of nature; and, (3) structural uncertainty, which reflects a basic lack of knowledge about the nature of the fishery system, its components, dynamics, internal interactions, including also the so called internal, behavioural and linkage institutional uncertainties [15]. The first two have been addressed quantitatively through a variety of analytical tools such as stochastic optimisation, behavioural models or simulation methods. For example, risk analysis has been increasingly used in practical settings to assess the implications of alternative harvesting scenarios in the presence of uncertainty arising both from randomness and parameter uncertainty. However, structural uncertainties are less tractable in the quantitative analysis framework7.

Structural uncertainties can produce shocks—unanticipated changes in the system—which may have major impacts on behaviours and management results. Taking into account the difficulties of facing structural uncertainties through analytical treatment, it

7 Typical structural uncertainties are the lack of information regarding the stock biomass, the species interacting in the fishery or the number of operating fishing vessels. Other instances of structural uncertainty are linked with spatial complexity, fish-fish interactions, fish environment interactions, technological change impacts and last but not least a high variety of institutional uncertainties. Institutional uncertainties may be grouped into three broad categories: internal uncertainties arising from the attributes of regimes themselves, multiple objectives, ambiguous objectives, institutional complexity; behavioural uncertainties arising from the actions of those subject to regimes-gaps between the ideal and the actual, compliance problems; and linkage uncertainties arising from spatial and temporal connections among regime-institutional overlaps, institutional change.
seems that they should be primarily addressed by designing management frameworks that will enable us live with uncertainty (i.e., a management framework able to provide acceptable results under limited or incorrect knowledge). Despite the fact that complete robustness is not possible, some management approaches, like adaptive management\(^8\) and also the precautionary approach, are relatively less sensitive to uncertainty.

Traditionally, the International Council for the Exploration of the Sea (ICES) has given fishery management advice on a *stock-by-stock basis*. But, recent problems in implementing this advice, particularly for the demersal fisheries of the North Sea, have highlighted the limitations of this approach. In the long-term it would be desirable to provide advice, which accounts for the inherent mixed-fishery effects of the North Sea. However, in the short-term, there is a need for approaches, such as the *mixed-species TAC approach* (MTAC), which can resolve conflicting management advice for different species within the same fishery, and generate catch or effort advice, which takes into account the mixed-species nature of the fishery.

Next to ‘guesstimates’, single species stock assessment, and the mixed-species TAC approach (MTAC), it is possible to apply more complex approaches, such as the precautionary approach and the ecosystem approach, into TAC setting. These progressive approaches point out that the knowledge base for fisheries management in general, and for individual right-based regimes in particular, tends to become broader, more complex, more multi-disciplined, tends to recognise more uncertainties and more mutual influences and tends to be more inclusive, incorporating fishermen’s knowledge. At the same time, they imply an increase in research complexity, implementation difficulties, the use of incomplete knowledge, and the integration of different types of knowledge. These different types of knowledge may prove to be conflicting, because of conflicting paradigms. For example, the precautionary approach has been taken as requiring that human actions are assumed to be harmful unless proven otherwise [16]. This most certainly does not coincide with the ideas and knowledge of fishermen. Such competing perceptions of knowledge make decision-making relating to TAC-setting more complicated. Thus, while there is a willingness and commitment to using different type of knowledge, it is important to consider which data is really used for analysing and advice; which data and knowledge is excluded and why; and who is to make that choice.

Although actually there is a stated willingness to incorporate knowledge from different disciplines and also fishermen’s knowledge, into the decision-making process of the TAC-setting; as above mentioned, the role of biological science and research-based knowledge is dominant in rights-based fisheries management. An excessive reliance on biological data has attracted criticism. To some extent, the inability of biological science to provide adequate information upon which to base stock management has been the reason for not anticipating population declines in several fisheries. In fact, there seems to be growing scepticism that fisheries can somehow be managed through the use of quantitative biological models. In the case of TAC-based systems, knowledge of fisheries

\(^8\) Adaptive management claims the need to continue learning about a fishery system over time in order to adapt to unexpected changes, which implies that new information should be integrated with existing information on a regular basis.
biology may not be sufficiently well-developed to provide the sole basis for setting TACs [17]. Based on the results of their empirical research, Batstone and Sharp [18] propose to use information contained in quota market prices as a guide to set TACs in order to complement the findings of stock assessment research in fisheries management. Under a rational expectation framework, their foundations rely on the acceptance that ITQ prices communicate information on future catches, incomes and profit expectations. From examining price formation in the New Zealand quota market, they found significant evidence that supports the use of quota prices to guide the setting of limits to commercial harvesting. Furthermore, time-series analysis provided a basis for studying the temporal response of changes in asset prices to perturbations in the TAC.

For another side, there is a link between fishermen’s perception of the resource and the legitimisation issue. In fact, when fishermen knowledge is not incorporated in the decision-making process or they disagree with the methods applied to generate the advice, legitimisation of a management system is difficult. In this scenario, a high probability of failure exists. Indeed, the failure of many attempts to involve users in the co-management systems in Africa and Asia can be associated with the fact that user participation was invited only regarding implementation issues, not with reference to normative or cognitive issues [19].

Another problem is according to Dengbol [20], “that fisheries biology is approaching the limits of cost efficiency relative to the value of fisheries and can still not deliver the goods in terms of numerical predictions”. Next to this, the models and concepts of fisheries biologists are becoming increasingly alien to fishermen and other stakeholders. A gap is growing between official science and other types of knowledge, like fishermen’s knowledge. “This gap is not just a question of lack of understanding or education on the side of fishers, but is rather associated with the basic scales at which the resource basis for fisheries is observed and understood”.

Next to the cost problem and the perception problem there is the so-called ‘chaos problem’: “there are principle limits to the predictability of any natural system beyond which it is impossible to assemble sufficient detailed data and models to provide any reliability” [21]. To incorporate knowledge from disciplines other than biology, including from users and stakeholders, in the production of knowledge for fisheries management increases the problems related to costs and chaos.

While new interdisciplinary and knowledge inclusive frameworks for robust, adaptive and precautionary management are important for living with uncertainty, there is also a need for appropriate management institutions to implement such a management approach. The existence of a non-governmental body with a focus solely on ensuring that conservation goals are met seems important (the allocation goals remaining in the hands of government). The Canadian ground-fishery offers an interesting example of changes in the decision-making process by the creation of the Fisheries Resource Conservation Council (FRCC), an independent organisation with a mandate to provide stock assessment and make public recommendations to the Minister of Fisheries and Oceans on TACs and other conservation measures. The FRCC is structured as a partnership between government, scientists and industry and was, in part, created as a result of a controversy over whether the Canadian Government had manipulated scientific information about collapse of the northern cod for political purposes.
3.3.1.2 From advice to political decisions: The institutional framework of TAC setting

The Canadian example illustrates that, in the real world, a TAC is far from being exogenous to the decision framework. But, who are the actors in the TAC setting process and what is their role within that process? Let us first begin by describing a hypothetical, but frequently assumed simplified model. Afterwards, specific real-life scenarios will be presented. Typically assuming an over-simplified model of social institutions, a scientific researching agency establish a TAC proposal incorporating the best self-validated knowledge, primarily following conservation criteria, and generally through the use of quantitative biological models, relying almost completely on the results of stock assessments\(^9\). Once the advice is on the table, the politician, who establishes the objectives and the hierarchy among them, takes the final decision: the one that maximises the benefits to the resources and users. The administrator will then be responsible for implementation of the TAC. Following the stakeholders’ map proposed by Mikalsen and Jentoft [22], the remaining definitive stakeholders are excluded from the decisions-framework process.

However, this basic model introduced above is missing important details. First, even if the stakeholders are outside the decision framework (although this is not always the case), they still influence the system through the political arena, especially if certain incentive-generation mechanisms (such as the ones driven by ITQs and/or cost recovery) are used. Second, stakeholders inside the decisions framework (scientists, administrators, and politicians) may pursue self-interests when advising, deciding and implementing. Those self-interests might involve minimising enforcement costs, maximising budgets, pursuing the best science, personal professional advancement, avoiding criticism, or simply getting re-elected [23].

It is thus important to discard simplistic institutional assumptions in order to obtain a good understanding of the specific processes, links and influences a particular TAC-setting approach may have. The Icelandic case is a good example within which to analyse the complexities and institutional uncertainties surrounding the decision-making process in TAC-setting, even when short-term decisions are subordinated to a long-term rule agree by the main stakeholders.

Decisions on TACs for individual species are taken by the Minister of Fisheries and based on the advice of the Icelandic Marine Research Institute (MRI). Assessments by MRI of the size of individual species are founded on data obtained from a variety of sources. First, the MRI leases a number of fishing vessels and crew and has them fish using their respective gear in the same predetermined areas year after year; second, MRI makes use of information from the catch logbooks of fishing vessels, landing reports and other data. The combined scientific data of the Institute is subsequently submitted to the

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\(^9\) Focusing on the dynamics of exploited populations, stock assessments describe the past and present status of fish stocks and forecasts their future trajectories for different exploitation scenarios. They includes studies of biological and population processes, such as growth, reproduction, recruitment and mortality. Typically, stock assessments include several related activities: biological research, the monitoring of resources through various survey techniques, the sampling of commercial catches, the assessment function (i.e., data analysis, population dynamics modelling, conducting risk analyses, reporting on resource status scientific peer review), the advisory function, and also the administrative function.
North-East Atlantic’s main scientific body—ICES. The final advice of the Institute is available 3 months before the start of the fishing season.

In 1995, the Icelandic government, upon the recommendation of MRI and with a wide consensus among stakeholders (specifically the fishing industry), adopted a long-term catch control rule (CCR) for cod, capelin and herring. In a broad sense, a CCR is a long-term law to set the annual TAC (or effort), usually based on the estimated state of the stock, recruitment and the TAC or effort in previous years, and tested with an extensive simulation framework\(^\text{10}\). The agreement of CCRs has important advantages for the complex institutional framework of influences and interests surrounding the fisheries management framework. In fact, a CCR alleviates the scientists of the task of advising certain catch rates rather than others, and gives managers a much firmer basis to resist heavy pressures to increase TACs.

The CCR for cod, which aims at looking for the fishing mortality that maximises current value of profit from the fishery by applying multi-species modelling and risk analyses, establishes that the TAC for any given year should be a constant proportion of the fishable biomass (concretely, 25%), a stage considered to be safe in the long-term, with a small probability of cod collapse. Soon after the adoption of the CCR, the catches per unit of effort (CPUE) started to increase and the stock to grow faster than predicted. Although fishermen alleged that during 1997–2000 the stock was being underestimated, during the next five years the CCR was applied as agreed. However, in 2000 the stock assessment showed a much worse state of the stock than was initially estimated. As a result, there was a sensible reduction of the TAC (by 60 kT), which was negatively received by the fishing sector. The Minister of Fisheries decided to put a maximum limit of 30 kT to the annual changes of the TAC, which effectively breaks the CCR. This systemic change led to very high exploitation ratios in the subsequent years.

The European Union (EU) framework, where the Council of Ministers has the final word in TAC-setting, constitutes an interesting example of the so-called political TAC. Based on data collected, processed, analysed and validated by ICES scientists from Member States, ICES’ Advisory Committee on Fishery Management (ACFM) reports on the status of the stocks and generates corresponding scientific advice for overall quotas. The European Commission has its own Scientific, Technical and Economic Committee for Fisheries (STECF) to comment on ICES’ TAC proposals. However, ICES and STECF are populated by scientists from the same research institutes. Except for economists (who are not part of ICES), the same scientists often sit on both committees. Subsequently, they are partly required to judge their own work. This situation highly strains the credibility of the STECF. Moreover, STECF has no budget, other than travel and subsistence expenses. Thus, its capability to produce original work is low. As a result, socio-economic advice on TAC-setting in Europe remains weak; more critically, the STECF is regularly consulted very late in the decision making-process and the Commission often takes little account of its advice \([24]\). Fisher leaders, NGOs and local authorities are aware of these weaknesses, of the role given to science in the CFP, and, finally, of the interests relating to the production of biological data \([24]\). Last but not least, fishermen and their organisations,

\(^{10}\) See for example, McAllister et al. for technical details of different approaches to be followed in order to set the CCR.
NGOs and other stakeholders often complain about being excluded from the TAC-setting process.

3.3.1.3 TAC setting: overall quota vs ITQ regimes

The key philosophy for overall TAC choice is to set a total fishing mortality level (F), which is a balance between what can be taken today and what should remain in the sea to grow and reproduce for the future. The application of this principle may vary according to population dynamics and the current status of the stocks under consideration. Undoubtedly, there needs to be sufficient research-based knowledge (both data and applied biological theory) to set appropriate and credible TACs. Analytical stock assessments are thus an essential requirement.

Although the technical procedure for setting the overall quota in an ITQ regime is basically the same as for any other TAC-based management system, there are several issues surrounding TAC-setting under an ITQ programme that are worthy of special attention. We refer here to the incentives favouring increasing demands of knowledge; and to increasing the participation of fishermen in the knowledge-generation process.

ITQs not only place increasing demands for quality on stock assessments, they also create further demands for socio-economic analysis. Besides the initial allocation and subsequent monitoring, socio-economic assessments are linked to the need to measure and evaluate the changes that individual quota systems may generate (including increasing profits, changes in the distribution of incomes, loss of employment, concentration of quota, population losses in fishing communities, etc).

ITQs are expected to generate pressures for greater involvement of ITQ holders in the scientific domain, because changes in TACs affect the value of their assets and, accordingly, their potential present and future profits [25]. While under overall TAC regimes fishermen’s individual incomes are particularly related to their skills to compete in the race for fish; under ITQs, incomes are strongly correlated to shares possession and, thus, become increasingly dependent on the overall TAC. It may then be concluded that individual quota holders are expected to have a growing interest in the TAC-setting process, from data collection, to even the methodology to set the TAC. In other words, ITQs may lead to an increase in industry’s participation in management issues in general and in researching concerns in particular [26].

Furthermore, if government requires that fishermen pay for the services that government has provided free in the past (if cost recovery is adopted), it may be expected that payers will want to ensure those services are obtained from the best sources and at minimal cost. Taking for granted that ITQ owners tend to group themselves into organisations, this raises the issue of their direct purchasing of researching services needed for fisheries management. Different arguments can be found in the fisheries literature when discussing the convenience or not of research-based knowledge liberalisation processes.

Following Scott [27], it is useful to compare a fishermen’s organisation’s TAC-setting committee and a governmental TAC committee: both under the best science incorporation assumption. The former’s final decisions will reflect the members’ private attitudes, preferences and expectations, which might differ from those of the governmental committee, which is expected to be guided by social preferences. Economic theory shows that the
social shadow value of resources is always higher than the sum of private shadow values and that fishermen are expected to be short-sighted (by discounting future harvests more heavily than present ones), especially if they have invested heavily and become embedded in the fishery. However, one may also argue that right holders are more interested in the future of the fish stock, because the consequences of making wrong predictions (and the rewards for being right) are personally grave for them. Last, but not least quota holders will be too little concerned with the survival of non-commercial bycatch species and broad ecosystems, which leads to the argument that governments should retain some control over important biological conservation parameters.

3.3.1.4 Recent developments in the use of knowledge in TAC setting: Collaborative research and direct purchasing of research

Since 1986, when New Zealand (NZ) put into practice a market-based approach to managing fisheries resources in the form of fully transferable ITQs, NZ’s fisheries management has set into a scenario of successive and interrelated events, tending to a growing participation of the so-called ‘definitive’ stakeholders in general and the harvesting sector in particular. Among others, those developments include cost recovery and the increasing efforts to separate science and policy within government. Contestable research (research projects let by competitive tender), collaborative research (the research that at some stage—initiation, conduct, and/or review—involves multiple stakeholders), and direct purchasing of research by the stakeholders, are inter-related approaches and phenomenon taking place in NZ market-based governance.

Cost recovery was introduced in 1994 as a method of funding all the services required for fisheries management (which obviously includes enforcement and researching costs). The charges are divided between the government (who faces the costs related with the recreational and customary aspects of each fishery) and the fishing industry, whose share is recovered through a levy system. This progress of the market-based system establishes a link between what fisheries management is worth to the industry and what it costs. The industry would be interested in maximising the value of the services that it gets for its money and in minimising what it pays for any given amount of services [28]. In circumstances where cost recovery is a feature of fisheries management, it is likely that the industry would demand, and in all probability get, increasing influence over management decisions. Evidence in NZ shows that cost recovery has generated further incentives for the ITQ holders to be significantly involved in both research and management. In fact, the levy system has been subject to much debate since its introduction, giving rise to discussion of general issues such as management efficiency and efficacy, and core questions such as who should supply the services or how these levies should be applied to quota holders.

One year after the introduction of cost recovery, most of the fisheries research staff within the Ministry of Fisheries (MoF) was restructured into the Independent Crown Research Institution (NIWA). The MoF retained a core of scientific expertise, and a policy advisory, regulatory and enforcement capacity. That restructuring, joined with a more transparent and ultimately contestable research planning process, is proof of the increasing efforts to separate policy and research aspects of fisheries management [29]. Since 1995,
the required research projects have been established through a meetings procedure open to all the stakeholders, and directed towards producing a white paper (listing all research projects required to manage the NZ fisheries in the coming year). Potential contractors are invited to submit their tenders, which must include full proposals on how to execute the work and a breakdown of the associated costs. The core scientific staff of MoF, in conjunction with outside scientific experts, evaluate and decide upon the contractor, who is required to report the results to MoF and to the pertinent stock assessment working group11.

It seems that the internalisation of the owner status of ITQ holders and cost recovery have driven incentives in the same direction. Industry move forwards to obtain increased participation in management and shows increasing willingness to achieve a better understanding of stock assessments as well as to contribute its own knowledge. These evolutionary processes in the cognitive pillar must be necessarily linked with the development and consolidation of quota owners’ organisations, another complement outcome of the quota management system (QMS) [4]. Depending on the fishery, these associations make fisheries management rules and impose sanctions for non-compliance on members, represent the interests of shareholders in consultation processes conducted by the government (such as the setting of the total allowable commercial catches—TACC), and facilitate the collection of funds to finance management activities (such as research). The umbrella organisation for these (about 30) associations is the New Zealand Seafood Industry Council (SeaFIC)12.

Collaborative research is a relatively new phenomenon that has spontaneously arisen as an indirect outcome of the incentives contained in New Zealand’s rights-based fisheries management regime, indicating a high level of cooperation among (biological) scientists, administrators and industry. This work is done in different working groups assessing the stocks. The main tasks of the groups are to estimate the level of sustainable harvest for each stock and to determine whether or not current TACs are sustainable. Based on the discussion in the plenary of all groups, an advice paper is prepared, on which the Ministry will respond with a new document, the Initial Position Paper (IPP). This IPP will be discussed again by the working groups and, the ministry bases the final decision on the outcome of this discussion [13].

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11 Eight stock assessment working groups—whose membership comprises MoF staff, representatives from commercial, customary Maori, and recreational fishing sectors, and environmental organisations—cover inshore, mid-water and deepwater fish stocks. Their main task is to estimate the level of sustainable harvest for each fish stock and to determine whether or not current TACs and TACCs (total allowable commercial catches) are sustainable. Three other working groups cover recreational, socio-economic and aquatic environment issues. The outcome of the fishery assessment plenary is released as an advice paper to the MoF, which makes the final decisions.

12 SeaFIC is a limited liability company owned by commercial fisher organisations, whose role is to represent the generic interests of the seafood industry, providing policy, support, information, as well as scientific, training and advocacy services, to the seafood industry. SeaFIC’s science unit employs its own fisheries scientists and has a number of collaborative research programmes with New Zealand and international research providers. SeaFIC hire experts to conduct analysis of stock assessments. They use the same data as the government-paid scientists of NIWA. They often come up with different results. SeaFIC has, at present, one permanent stock assessment expert and a number of experts on short-term contracts, all participating in stock assessment meetings.
The biologists participating in this collaborative research process do not all originate from the NIWA; biologists hired by the industry also participate. “Devolving research to stakeholder groups and in particular to quota-holder groups was a strategic decision made in 1997 by the then fisheries minister and government” [13]. According to the biologist Starr, who participated at both sides of the table (as Ministry representative as well as industry expert) the benefits from this contested process are several: intense peer review leading to better assessments; development of new methods and data sources; and industry consultants relieving government scientists of some of their workload by conducting part of the stock assessment analysis. By buying expertise, the fishing industry is also buying itself into the system of scientific advice, which could be an alternative to lobbying. From the seafood industry viewpoint, the collaborative research not only contributes to the sustainable use of fisheries, but also confers a comparative advantage on New Zealand’s seafood industry in international markets. Furthermore, it provides increased certainty to guide long-term investments. However, some groups belonging to the industry only have eyes for the cost side of the research and still consider topics such as impacts of fishing on ecosystems and biodiversity as environmentalist strategies to close fisheries [30].

The Chatham Rise Hoki catch sampling programme is an example of industry-led research. Concerned that there was insufficient management-related information, the New Zealand Hoki Management Company (HMC) and SeaFIC initiated a sampling programme that involved measuring and sexing the fish and assessing the condition of the gonads. All the vessels operating in the fishery were included in the sampling programme, which was undertaken by the crew-members themselves after a training period. The scale and coverage of this sampling is bigger than previous government-run sampling programmes, thereby resulting in a more robust data set that enables industry and scientists to have more confidence in the results of associated analysis.

Even more industry-driven initiatives have been conducted by The New Zealand Rock Lobster Industry Council (RLIC)\(^{13}\). RLIC actively participates in the multi-stakeholder National Rock Lobster Management Group (NRLMG), a primary source of advice to Ministers on all matters related to rock lobster fisheries, whose membership includes government agencies, commercial, recreational and indigenous fishers’ representatives, environmental non-governmental representatives and scientific advisers. In 1997, RLIC became an accredited research provider to the Minister of Fisheries (MoF), and, since then, has successfully tendered for, and executed, several rock lobster stock assessment contracts. Research contracts are undertaken in collaboration with national science providers and internationally recognised stock assessment consultants contracted to RLIC.

Until recently, although all stakeholders were free to purchase or carry out their own independent research, fisheries research was purchased by the Ministry of Fisheries, who, in consultation with fisheries stakeholders, determined what research was to be

\(^{13}\) RLIC is an umbrella organisation for nine commercial stakeholder organisations (called CRAMACs), established as incorporated societies or limited liability companies operating in each of the rock lobster management areas of New Zealand. CRAMACs are shareholders in RLIC and assign the nine-person board of directors, one from each CRAMAC. Membership of CRAMACs comprises quota owners, processors, exporters, and fisherman in each region. Governance is based on a two-tiered voting procedure that gives priority to quota ownership on issues affecting TACC decisions, and certain government consultation processes.
Right-based fisheries management

done and where. However, industry organisations such as the RLIC and the SeaFIC have put together research proposals and won contracts for the delivery of the required researching services. The main motivation for this step is both to be more closely involved in researching, to have a better control of research costs, and to introduce new stock assessment and research methodologies (such as Bayesian stock assessment models). However, in the mentioned cases, industry organisations are not the direct purchasers of the research services, they are the providers (i.e., contracted by the Ministry of Fisheries).

An amendment to fisheries legislation allowing the direct purchase of required fisheries services by stakeholder organisations can be interpreted as a step to further liberalisation of the researching provision services. Any fisher organisation, subject to meeting appropriate standards and specifications, is able to purchase and fund research directly, although directly purchased research is still determined by the research planning process described above. However, progress on direct purchase was slowed by opposition from environmental NGOs and some scientists, and also by the reaction of some segments of the fishing industry. The former argue that, since fisheries resources and also researching services are public goods, research should be a core responsibility of government. Furthermore, they maintain that industry may have incentives to distort the results of research or pressure contracted providers for short-term gain. In contrast, some harvesters’ perception is that direct purchase means devolution of management responsibility, rather than delegation of the MoF research-purchasing functions.

This evolution in the knowledge generation and validation processes taking place in the framework of the NZ quota management system indicates the following of a market-oriented path and a parallel increasing participation of seafood industry organisations, particularly the ones related to the most valuable fisheries. To a certain degree, Townsend’s [23] ideas of corporative fisheries management are gradually invading the normative, regulative and cognitive pillars of the NZ fisheries governance. Remember that under such a hypothetical regime the corporation would face almost all the issues related to fisheries governance. The state’s role would be minimised, just acting as the negotiator and enforcer of the operating contract, regulating the negative effects of fishing activity upon resources not owned by the corporation.

Away from New Zealand, in the Canadian offshore, economically successful, sea scallop fishery, quota-holders collaborate closely with government managers and scientists through an informal co-management agreement with the purpose of stabilising harvesting strategies according to the scallop’s fluctuating recruitment patterns. Industry not only finances most of the standard science, monitoring and enforcement costs of this fishery, it invests millions of dollars in a government research programme to determine an accurate and precise estimate of recruitment abundance. Using sophisticated benthic sonar mapping technology and sample surveys, together with daily harvesting data, scientists are able to determine the location and abundance of specific year classes of the sea scallops in later years [31].

14 Public goods exhibit two main characteristics. (non-rivalry in consumption and non-excludability). Information and researching services, once generated, can be used by anyone without reducing them and no one participating in the industry can be excluded from enjoying the benefits. See Arnason et al. [28] for more details.
Some examples of collaborative research can also be found in Europe. For example, in France the IFREMER-PROMA partnership on deep-water species manages the majority of the associated French quotas. The Producer’s Organisations (POs) hired a biologist on the basis of a collaborative agreement with the national fishery research centre (Ifremer). The PO’s objective in internalising scientific expertise on deep-water species was to improve the quality of catch data, to build-up trust between scientists and fishermen, and to make use of fishers’ experience and knowledge [24]. The North Sea Commission Fisheries Partnership, established in 2000, in 2002 resulted in a more formal cooperation between fishermen, industry representatives, and independent scientists to discuss some of the ICES preliminary stock assessments. These stock assessments result in TAC advices that will be divided among Member States as national quotas and, in some countries like the Netherlands, as ITQs. ICES scientists now receive information on the status of North Sea fish stocks, which has been collected by fishers. “This working group is a real break-through as it gives us the opportunity to work closely with fishers and use their logbook data to help us build up an even better picture of the status of North Sea fish stocks” [32].

In summary, all these examples are indicating a new development in the use of research-based knowledge. One could call this the privatisation of knowledge production and a more collaborative way of producing knowledge, using mandatory scientific knowledge as well as industry financed knowledge. This willingness and attempts to use data of fishermen indicate a furthering of the collaborative process between scientists and user-groups.

3.3.2 The knowledge base for ITQs: efficiency and welfare objectives

Any property right, as well as any subsequent transactions that may occur, needs to be registered. This requirement applies to individual quotas and entails the bureaucratic production of knowledge. Accordingly, to manage a QMS a great deal of information is required—on individual landings and/or catch data and on quota transfers. ‘Everything’ needs to be registered in logbooks and in auctions, and this will become the input of automatic, digital information systems. This information will be transformed into aggregated fleet data, from which one derives statistical knowledge.

With ITQs, the necessity for socio-economic analysis is multiplied. In addition to establishing the initial allocation and fulfilling monitoring purposes, socio-economic assessments are linked to the need to measure and evaluate the changes that an individual quota system may generate. Knowledge on the operation of the market, fleet structure changes, changes in the number of fishermen, quota prices, and actual quota-uptake are of vital importance to an ITQ management system.

3.3.2.1 The knowledge base for the initial allocation

Once the TAC has been set, it has to be divided into individual quotas. This process determines the initial welfare distribution of the system. Usually, quotas are allocated free of charge to vessel-owners, mainly on the basis of their recent catch history, following not only to replicate the standing fleet structure, but also the acceptance of the fishing
industry. Thus, empirical evidence shows that individualised catches from 3–5 years prior to the quota adoption and, in some instances, individualised data of vessel characteristics and recent investment levels, form the data requirements for the initial allocation. Where individualised data on past catch performance is poor, or may be subject to manipulation, the initial allocation process may not only result in much litigation in the courts but also spread mistrust and conflict among fishermen, fishing communities and industry.

Although individual quotas are, in fact, exclusive use rights, they are often considered to be pseudo property rights. A, seemingly obvious, aspect of property rights is that the owner actually knows what exactly it is that he owns. However, the holder of ITQs does not know in advance how much fish the ITQs represent. This is because, in the majority of cases, quotas are a share of an undetermined and fluctuating quantity (the TAC), only specifiable less than one year in advance. Thus, whatever the TAC set at the beginning of the year, it is still possible in most cases to alter the TAC—and thus the ITQs—if sudden changes occur to the stock. For example, in Iceland, even at the beginning of the year, the precise tonnage represented by the ITQ is not certain [8]. In New Zealand, the initial allocation proved to be an even more complicated affair than setting the TAC. The decision was to allocate according to catches of the average of the two best of the three preceding years. Based on these statistics, 1800 fishers received their provisional allocations in 1985. However, fisheries statistics at that time was not an exact science and quite a number of fishers had, for many years, dodged the tax authorities by systematically underreporting their catches. In the end, 1400 reviewed their initial allocation, causing further delays in the process.

The introduction of IQs in 1976 in the Dutch flatfish fishery required knowledge about track records and information about the fleet structure and data on enterprise level. A limited part of the national quota was not included in the allocation, but was kept as a 'National Reserve'. This reserve was meant to compensate for eventual excess landings. The quota was allocated on the basis of historical catches and/or engine power. The individual quotas received by fishermen fishing prior to the 1st of January 1974 were based on the highest amount of plaice and sole landed in the years 1972, 1973 and 1974. For ships under 1250 hp commissioned after this date, quotas were based on the average performance of the vessels in the same hp-group. For ships with more than 1250 hp, quotas were fixed by the Ministry. This system met a lot of resistance from parts of the industry because it resulted in considerable differences in quotas between vessels of similar capacity. As a result, the system was revised in 1977 (the 1977 allocations are still the basis of the present quota system) adjusting IQs both to engine power and to historical performance. In the 1977 system, fixed by-catch quota of flatfish per hp-group for non-beam-trawlers over 250 hp were frozen at the 250 hp level. This resulted in a relatively large number of small quotas. Because of their limited size, these quotas are often referred to as ‘mini quota’.

The most significant step in the introduction of ITQs in Icelandic fisheries was taken in the Fisheries Management Act of 1983, which regulated fishing in the Icelandic EEZ during 1984. In this Act, it was stated that fishing for the most important groundfish species (cod, haddock, saithe, redfish, plaice, Greenland halibut and catfish), by vessels over 10 GRT, should be regulated by transferable vessel quotas. The quota allocation was mainly based on historical catches in the period November 1st 1981–October 31st 1983.
Together with herring and capelin, these stocks constitute more than 90% of the value of the catches of Icelandic fishing vessels [8].

Apart from this knowledge of track records and vessel characteristics, data on recent investments at enterprise level and/or fleet structures are also required for allocating initial quota. This allocation is a form of welfare distribution that also needs to be underpinned with social considerations and framed within constitutional law.

3.3.2.2 The knowledge base for transferability rules

By definition, an ITQ holder can transfer his quota to someone else. Transfers can be made on a permanent basis, as a form of sale, or on a temporary basis. Many temporary arrangements are possible: temporary use, swap, short-term lease or long-term lease. However, some boundaries and restrictions can be set, often to avoid some undesirable distributional consequences, such as the concentration of quotas or changes in the fleet’s structure. Any quota transfer must be recorded.

For example, in Iceland the Fisheries Management Act of 1990 allowed trade in both quota shares and annual quotas, but under some restrictions. Within this system, it is, first, required that all quota shares are registered on a specific vessel and that the quota shares that are registered on that vessel do not exceed its capacity. Second, if trade in quota shares implies transfer of quotas from one municipality to another, the municipal council where the seller is registered has the first option to buy it. Third, a single firm is not allowed to hold quota shares for cod in excess of 12%, redfish in excess of 35% and for haddock, saithe, Greenland halibut, herring, capelin or off-shore shrimp in excess of 20%. It is also prohibited for one firm to hold aggregate quota shares in excess of 12%. Finally, vessels not catching at least 50% of their quotas allocated to them in one of two consecutive years lose their quota shares [8]. Trade in annual quotas in Iceland is also subject to limitations. It is not permitted to lease (net) in one quota-year more than 50% of annually allocated quota. Additionally, under the terms of the Fisheries Management Act of 1990, the transfer of annual quotas to a vessel in another municipality could only be approved by the Ministry of Fisheries after consultations with the municipal council in question and the board of the local seamen’s association. There are hardly any examples where this procedure prevented some trade in annual quotas. This was changed in 2001. The present law requires that the Directorate of Fisheries receive confirmation from the Price Board for the Share Value that there exists an agreement between the vessel-owner and the crew on the landing prices that should be used for deciding the crew’s share before agreeing to the transfer [8].

In the New Zealand case, there had been serious concerns both about the regional consequences and the aggregation effects of ITQs. Finally, regional limitations were not accepted, on the grounds that supporting certain areas would only handicap the quota owners in the same area: “locking them up” in less valuable markets. Fear of excessive aggregation was addressed through quota aggregation rules, generally stating that no single owner could obtain more than 20% of the total TAC of a species in any quota management area (QMA), except rock lobster and paua (10%) and some deep-water species (35%). Selling, leasing or even sub-leasing was permitted from the start and trades had to be registered with the Ministry. At first, trading was performed through the Quota
Trading Exchange (QTE), but this was only in operation for two years. Among other reasons for this, quota holders claimed that the QTE revealed too much information to the Ministry, later to be used to adjust resource rentals. By 1990 80% of quotas had already changed hands—an indication of the speed of the restructuring of the fleet, according to Hersoug [13].

When, in 1976, the quota system was introduced in the Netherlands flatfish fishery, quotas were only formally transferable together with a vessel. Soon, however, it proved to be possible to circumvent this rule by using legal constructions, and a spontaneous quota market came into existence. In 1985, quota became officially transferable independently of vessels. The transfers were subject to the following rules: quotas can only be bought by owners of a fishing vessel that is registered on an EU list and who are in the possession of a licence; fishermen can only sell their plaice and sole quotas as a total whereas it is allowed to buy parts of these quotas; the transfers have to be approved and registered by the Fisheries Directorate. Since 1985, the transfer of quotas has been subject to rules restricting that transfer to limited periods during the year. This was introduced in order to prevent doubtful transfers at the end of the year when quotas are nearly exhausted [33]. Since 1993, the ITQ system has overlapped with a co-management system, in which fishermen pool their quota in quota management groups. The groups register every quota transaction (selling and renting) and the quota uptake. Landings are registered at auction and there is a voluntary agreed auction duty for transparency sake. Logbooks of vessels and the VIRIS system (a catch registration system) are used to check on quota uptake during the year. Fishermen can check on the websites of their groups and organisations for quota uptake percentages during the year and are warned when they have nearly consumed their annual ITQs. At the beginning of each year, quota holders receive a letter from the Ministry in which it is indicated the number of kilos they are entitled to fish for each species, depending on the annual TAC/national quota and on the percentage of the national quota the fisherman owns.

3.3.2.3 The knowledge base for distributional objectives under ITQs

ITQs have usually been allocated free of charge. Over time, rents have emerged in the industry and become capitalised into a market value for quotas. Some quota owners have made a windfall gain from having obtained fish quotas for free and sold them on. The ongoing controversy over ITQs in Iceland is in part this intergenerational equity problem. Many people object to the fact that fishermen who were initially allocated some quota for free can sell it for a profit. These objections have become stronger as quota prices tend to be very high to the extent that the quota price tends to exceed the unit profit. This makes the pure profits of new entrants negative [34, 35]. ITQs may also aggravate financial succession problems in family companies, which many fishing companies still are. In the Netherlands, fishing sons of quota holders have had problems taking over the fishing enterprise because ITQs became so valuable. Fathers cannot simply give the quota to their sons because of the Dutch tax system (law of inheritance and capital transfer tax).

Capital building on user rights is not a unique feature of ITQs. With other management options applied to reduce fishing effort transitional gains will also occur, though with
ITQs it may appear more explicitly. For example, in the case of IQs the value of the right will be hidden in the vessel price creating similar advantages and disadvantages as those mentioned above for the ITQs [8].

To deal with windfall gains, the quota rent could be confiscated in one way or another. However, some economists argue that such resource rentals would retard economic growth, as more of the rent would be invested profitably if it remained in the hands of the quota owners. Moreover, taxing fishery rents would reduce the value of the quotas and so weaken the associated common interest in good resource management.

In Namibia, the fishing industry pays fees for fishing licenses and quotas amounting to about 8% of the landed value. In the British Columbia halibut fishery, there is a resource rental fee per tonne of quota, and, in addition, the industry pays the costs of managing the quota. Iceland has recently introduced modest resource rentals, which will not do much better, however, than cover the management costs of the fisheries. But some countries have elected not to apply resource rentals. New Zealand abandoned its original plans for resource rentals as a part of a dispute settlement with the industry when the share quotas substituted the initial fixed quotas. In the United States, it is illegal to claim resource rentals, but the industry can be charged for management costs of ITQ systems.

Even after the stocks have recovered and they are managed optimally so that they give greater catches than today, technological improvements will lead to decreases in employment in fisheries. Fisheries cannot, therefore, be the backbone of a growing community, except in the case where that community has the capacity and opportunity to increase its share of quotas. As the total quota has to be limited, such a development would inevitably mean that some other fishing communities will have to lose out. For those communities—and for fishing communities generally—regional policies should be directed towards creating employment opportunities in industries other than fishing: diversification. Economic analysis and experiences in some countries indicate that restrictions on the transferability of quotas and subsidies for purchases of quotas in some regions will only make it more difficult to increase the fishing activities in those communities that are best suited for this industry. In an ITQ-system such practices lead to higher prices for the quotas, making it more expensive for successful fishing firms to grow and making fishing in general more expensive and, thereby, less profitable [8].

In order to have knowledge about the realisation of distributional objectives, it is necessary to evaluate the distributional effects of each ITQ system in each specific situation. This will entail long term socio-economic analysis of changes in the community structure, of the costs and earnings of enterprises, of the asset and liability structure (balance sheets) of fisheries companies, of the industrial structure (ownership, vertical and horizontal integration of the fishing industry, labour, and other contracts, etc.), of commodity flows, input and output prices, and of fishing effort and the use of inputs by fishery and fleet segments, as well as the nature of the fishing fleet and processing capital.

3.3.2.4 The knowledge base for enforcement

For a fishing right to have any sense it must be enforceable. An individual vessel owner needs guarantees that he will be able to catch his owned quota, otherwise there would
be no incentive to buy that quota. Enforcement is usually achieved through a system of monitoring and the prosecution and punishment of encroaches. The successful operation of an ITQ programme requires that the monitoring system be seen as capable of detecting abuse. Penalties for non-compliance should be firmly established, rigorously enforced and severe enough to encourage compliance.

ITQs, comparing with overall quota regimes, multiply the enforcement requirements. Global TAC programmes must have a way to monitor the total harvest and ensure that the TAC is not surpassed. In addition, in the case of individual quota systems (IQs), it will be also necessary to monitor the harvest of each participant to guarantee that it does not exceed their annual harvest rights. Furthermore, if quotas are transferable (ITQs), it is also essential to keep track of the current amount of share rights and annual harvesting rights owned by each participant.

In practice, ITQ systems require rigorous record keeping and mandatory reporting and disclosure rules. Although dockside verification of landings will still be important, ultimate success will require computerised systems of reporting and data management. Additionally, there should be at least two sources of information for any transaction. Trip-by-trip monitoring is also advisable, especially where larger and sophisticated vessels are involved. Multi-species fisheries with difficult by-catch problems require the use of at-sea observers to deter discarding. Some at-sea monitoring requirements can be aided by new technologies. For example, satellite tracking can be useful where specific fishing areas have to be monitored. Increasingly, new computer and communication technologies permit information to be maintained in electronic logs while vessels are fishing and allow these data to be transmitted in real-time, thus facilitating monitoring of the fishery and fishing operations.

But, the most brilliantly designed management system will fail unless the fishers obey the rules that support it. The success of a fisheries management regime depends on achieving the highest possible levels of compliance with the rules that underpin it. More than just enforcement is needed to achieve high levels of compliance. You must have the fisheries stakeholders with you [36]. Resource users are far more likely to accept a system when they see it as having legitimacy in terms of outcome and process. The way to achieve this is through stakeholder participation. To work in practice, the rules of the management system, and the services that support that system, must be developed and operated in collaboration with the “regulated community” and other stakeholders.

In New Zealand the monitoring system is based on the fisheries record and return system known as the documentary product flow system [36]. It creates a paper trail relating to the movement of fish and fish products through the marketing chain. The aim of the system is to provide valuable fisheries management information and allow for the detection of quota-busting. The system tracks all paperwork associated with the flow of fish and related financial transactions from harvester to first point of sale and to other dealers and retailers. Quota owners have to authorise harvesters to catch fish against their quota, keep business records, submit monthly catch-against-quota returns, and register the purchase and sale of quota. Harvesters must have permits to land their fish at designated New Zealand ports, keep records of catch, fishing effort and landings and submit monthly returns. The main control point in this system is the first point of sale of the fish from the harvester. This first point of sale, or fish receiver, must be licensed,
keep business records, submit monthly purchase and sale returns. Dealers in fish and retailers (second or subsequent points of sale) are required to purchase fish only from licensed fish receivers and keep business records. The compliance monitoring system is based on carefully matching the catch and landing returns supplied by harvesters, the returns of fish purchases and sales by licensed fish receivers, and the catch against quota returns and quota trading documentation supplied by quota owners. The compliance monitoring system also includes a satellite-based vessel monitoring system, an observer programme, a licensed fish receiver system audit programme and a harvester/licensed fish receiver/dealer fish inspection programme.

The maximum penalty open to the New Zealand courts for each contravention of a serious fisheries law is a fine of $NZ250,000, the forfeiture of property used in committing the offence (for example, vessels, gear, cars, etc), and the forfeiture of illegal fish and quota. In practice, most monetary penalties have been relatively low. However, the ‘true’ penalty has been the forfeiture of the vessel and quota.

3.4 CONCLUDING REMARKS

Assuming good enough knowledge to set the appropriate TAC, the initial allocation and transferability rules, and also to face enforcement, ITQs seem to create a mechanism for increasing efficiency in fisheries and an instrument to remove excessive capacity. However, ITQs are criticised for a number of reasons. In particular, for the incentives they might create to discard, for the inequitable gains to those that received the initial allocation of the quota coming from a public good, and for creating regional imbalances and the demise of fisheries-dependent regions. Although, in general, the empirical evidence shows that the efficiency objective seems to be satisfied, the goals relating to ecological conservation and welfare are less fulfilled.

As catch quota systems create incentives for discarding, alternative market based formulations might be considered. For example, since 1996, the Faroese have combined a spatial and effort-based system of fisheries management. Two main aspects of the current fisheries management system in the Faroe Islands are of particular interest. First, an effort quota system for most demersal groundfish gear types, based upon individual transferable effort quotas (ITEQ) is in place. Second, a spatial and temporal closure system applicable to all or selected gear types is also in operation. Particular measures include temporal all-gear closures during spawning periods, as well as permanent spatial closures to trawl-gear types [37]. Actually, effort quota systems might be a solution in multi-species fisheries if fishermen cannot technologically ensure the composition of their catches or if fishing for the different species is equally profitable. But, if profitability varies, managers may want to limit catches of individual species, which would require some kind of catch limitations. But, the effort quota system has other drawbacks compared to the catch quota system. The most important ones are the difficulty to predict the fishing mortality (catch) from a given fleet capacity, the difficulty of controlling the total effort when new investment and new technologies are introduced, and inferior incentives for increasing the value of the catch through handling, storing (including the length of the fishing trip) and on-board processing [8].
Our knowledge regarding size and growth of the fish population and the behaviour of fishermen is limited, and such information is imperfect. This means that the optimum biological or bio-economic TAC cannot be precisely determined in order to achieve maximum efficiency and/or conservation goals. Thus, ITQs are expected to efficiently reduce fishing effort, but they will generally not reduce it to exactly the optimum level. Even when data availability is sufficient, due to uncertainty the biological and/or bio-economic TAC could be inappropriate. Additionally, where individualised data on past catch performance is poor, or may be subject to manipulation, the initial allocation process may not only result in much litigation in the courts but also spread mistrust and conflict among fishermen, fishing communities and industry.

Whether ITQs can be implemented as a management tool may depend on the socio-cultural and political circumstances of the country. ITQs may be politically incorrect. They may have very high political costs if they are implemented without the acceptance of stakeholders. *Ex ante* and *ex post* controversies resulting from opposing interests may undermine or inflict change on quota systems, may provoke environmentalist concerns, or provoke community demand for group quotas. These socio-political outcomes are dependent on countless factors, such as management goals, social preferences, physical characteristics and situation of resources, structure of the fishing industry, history and cultural traditions, share of the fishing in the gross income, attitudes of the fishermen and stakeholders, management costs, and last but not least the existence of *good enough accepted knowledge*. Reliable biological, ecologic and socio-economic fisheries statistics are not enough. Efforts should be made to incorporate and complement different types of knowledge. In fact, based on some of the earliest ITQ real experiences we have shown an increasing user’s participation in the knowledge generation and validation process. In fact, these recent experiences show new paths in the use of research-based knowledge: a tendency towards the privatisation of knowledge production; or a more collaborative way of producing knowledge, using mandatory scientific knowledge as well as industry-financed knowledge. Next to this willingness and attempts to use data of fishermen indicate a furthering of the collaborative process between scientists and user-groups.

Effective fisheries management is really about managing people living and working in a specific and always complex institutional framework; it is about influencing the behaviour of stakeholders to help achieve society’s objectives (whatever they are) and allocation goals for its fisheries. If fishers disagree with the normative aspects (such as welfare objectives) of management or have perceptions that differ from the cognitive base used for management decisions (such as research-based stock assessments), their voluntary compliance with the regulative measures or even with the objectives being achieved cannot be expected [38]. One way to generate the correct incentives is to incorporate the knowledge of at least the primary stakeholders. But, even if there is agreement to do so, this is not a trivial mission due to the perception problem. Some simple and agreed meta-indicators could serve as a communication bridge between different kinds of knowledge. One of the major challenges is to reach consensus among the stakeholders on relevant meta-indicators, which requires negotiation and avoiding an exclusive biological-science base.
REFERENCES

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Chapter 4

The Knowledge Base of Co-Management

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4.1 INTRODUCTION

Approaches based on decentralisation and bottom-up style fisheries management are becoming increasingly popular and are thought to hold the key for the sustainable exploitation of marine resources. It is in this context that participatory management and co-management are referred to [1]. Decentralisation of fisheries management authority from national to lower levels of management is on the political agenda in many coastal states in Europe, North America and in developing countries, often accompanied by calls for greater user participation in management [2] and, in fact, a recent review suggested that several hundred cases at varying spatial scales, currently exist [3].

When adopting a co-management system in fisheries, there is a need to move away from the traditional model of the development of the knowledge base for fisheries management, where knowledge is dominated by scientific research. This is not to say that scientific research is wrong or inappropriate; instead, it is to say that, by failing to incorporate the knowledge that resource users have, gaps in the knowledge base may result. If this is the case, management plans based on an incomplete understanding of the resource are, ultimately, flawed.

Fishers can be a valuable source of information and may have detailed knowledge of traditional and current patterns of exploitation and consumption, and can provide insight into potential problems with management plans [54]. One of the things that most sets fisheries apart from other areas of environmental dispute, such as pollution, is the degree to which the interested public, i.e., the fishers, see themselves as having sufficient knowledge to directly dispute the findings of professional scientists. Research can then be carried out collaboratively between resource users and scientists to gain a more complete understanding of the resource and, from there, to bring about rehabilitation of overexploited fisheries. Collaborative research takes on different forms with varying degrees of input from stakeholders and scientists.

A major challenge of co-management is the construction of a shared knowledge base of scientific and users’ knowledge that is accepted by government. The main question
raised in this chapter is: What is the knowledge base for co-management? Or, rather, what should be the knowledge base for co-management since there are only few examples of what Pinkerton [4] calls ‘complete co-management’. To try to answer this question we will first discuss the problem from several angles. First, we offer a brief general discussion of co-management and the attractions it holds for the different participants. Then, we will look at the different forms that the knowledge brought by participants to the co-management process can take. This is followed by an examination of the influences of differing world views on knowledge and introduce a “democracy paradigm” as a way of understanding the way these different world views come together dynamically. Finally we examine four different models of creating a knowledge base for co-management through collaborative research and other knowledge-focused actions.

4.2 THE IDEA OF FISHERIES CO-MANAGEMENT

Co-management is a broad concept and the context in which it occurs is very diverse. For this reason, the concept of co-management needs some flexibility. A recent definition of co-management is:

Co-management is a collaborative and participatory process of regulatory decision-making between representatives of user-groups, government agencies, research institutions and other stakeholders [5].

Essentially, it is about partnership and power-sharing in management of in situ fish resources. The most powerful reason for adopting co-management is the failure of governments to effectively manage capture fisheries, resulting in overexploitation [6]. There are three principal groups of stakeholders who must be considered: these are governments, user-group communities and the scientific communities. When other stakeholders, notably conservationists, become involved it is becoming common to refer to this as cooperative management as distinct from co-management.

The reason for this movement towards co-management is that it has become increasingly apparent that the top-down approach to fisheries management has failed to prevent overexploitation. This is, in part, attributable to the non-inclusion of fishing communities fully in the management process, leaving them feeling isolated from the whole management process, resulting in a growing gap between the resource users and the regulatory authorities and scientists. Although co-management seems to be a rather new concept, this is not at all the case. Throughout the world, co-management systems in fisheries already existed before the term was in use. Co-management-like arrangements in fisheries can be found under other names, such as the Prud’homme in France, the Cofradías in Spain, Mazoperias in Poland, and the Sasi in Indonesia. Further, practices of co-management occur in different social, cultural and ecological settings in Europe, New Zealand, the USA, Africa and Asia, and are not strictly restricted to fisheries. What is now called co-management often originates from older institutions. In fact, corporatism
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and neo-corporatism in Europe has some of the features of co-management and can be found in sectors other than fisheries, such as agriculture [7].

Co-management, unlike community-based or self-management, does not challenge the authority of the state. Instead, co-management draws upon the resources of both society and of market institutions [8]. Indeed, co-management has been shown to produce synergies in which both the local and government aspects of environmental protection become stronger simultaneously [9]. The concept of co-management was developed from the notion of democracy, promoting social equality by sharing the power of the government with the people being governed [10].

The rationale for co-management contains three main elements. First, the resource users have an in-depth knowledge of the fishery, gained from experience, which can be added to the information attained by fisheries scientists. They have knowledge of traditional resource use, historical levels of resource consumption and may be aware of any problems of implementing management regimes [54]. Second, involvement of the community encourages compliance because they fully understand the policy and why it was created, which leads to commitment and support of the policy from the resource users [54, 11]. Third, democratic theory would imply that those who are to be affected by a management decision should have their say [12].

One of the principal motivations of the state to partake in co-management is to gain the help of fishing communities with aspects of fisheries management that require local knowledge not at the disposal of the state [13]. The dilemma for government when considering the involvement of stakeholders is the expected increased complexity in a democratic decision-making process. Fishermen are motivated by conflict resolution, increased funds that may become available, by declining stock sizes coupled with increased competition, and a longing for influence by the recognition and use of their knowledge.

In the Netherlands, for instance, an important motive for individual fishermen to join co-management groups was to improve the relationships between different groups of quota-holders, as well as the improvement of their relationship with the government [7]. For example, a group of Dutch fishermen share data on catches with scientists in the so-called ‘F-project’. Fishermen are also concerned, however, with potential costs, risks, increased responsibilities, and the fear of heightened conflict levels. Devolution from one administrative level to a lower level does not only affect the vertical dimension, it has consequences for horizontal linkages. This is particularly the case when user-groups and stakeholders are allowed to play a role in fisheries management. Devolution may affect their relative power, as some groups are much better organised, and hence more capable of participating effectively, at one level than they are at another level. If for instance, fishermen are better organised at the national level than at the local level, they would resist devolution, especially if they are up against other stakeholder groups that have a strong local representation. Users and stakeholder groups may also fear a more complex management system that is more difficult to oversee, within which they would be subject to additional and geographically incoherent rules and regulations, particularly if the cross-scale and linkage issues are not properly addressed. [14, 15].
The scientific communities are motivated by the growing knowledge gap between scientific knowledge and local knowledge, although they are hesitant about the reliability of the users’ knowledge base. The fisheries research discourse developed, following Dengbol [16, 17]:

In response to the emergence of international fisheries management institutions and has developed through the 20th century in response to emerging fisheries management issues. The major break in this development was when the unit of observation changed from being processes on the local scale to the ‘fish stock’ averaged over larger scales, around 1920. This coincided with the internalisation of fisheries management and ‘rational fishing’ emerging as the main management discourse, away from an approach focusing on economic development and modernisation. (p47)

Adaptations, including optimisation and precautionary approaches, occurred within the same paradigm of rational predictability of the outcomes of management on a fish stock basis. By adapting a large-scale averaging approach, fisheries research has, at the same time, alienated itself from the observations and understandings that are associated with commercial fishing activity, where predictability with a high resolution in space is required. This has led to a loss of legitimacy among fishers of knowledge created through fisheries research.

The fisheries research discourse will also approach cost limits as it is attempting to internalise more complex processes and systems (e.g., ecosystem approach) and stochasticity within the same predictability paradigm” [16, 17].

4.3 DIFFERENT FORMS OF KNOWLEDGE

An important dimension of the problem of formulating a knowledge base for management within a co-management context is that different stakeholders hold knowledge about the fishery in different forms. Three main distinctions can be made, that between tacit and discursive knowledge, that between written and oral knowledge, and that between anecdotal and systematic knowledge. There are important interplays between the forms and the content of knowledge relevant to fisheries and control over both the form and content of knowledge is critical in any political process. Fishers, in general, see themselves as having sufficient knowledge to directly dispute the findings of professional scientists.

4.3.1 Tacit vs. discursive knowledge

Tacit knowledge is knowledge that people have, but that is not (easily) expressed. Discursive knowledge means knowledge that is shared and expressed. The question of tacit knowledge plays an important part in general discussions about institutions and knowledge. Some fisheries management scholars, notably Gísli Pálsson [18, 19], have argued persuasively that much of the knowledge that fishers have of the resource is tacit knowledge. How well fishers’ knowledge can be articulated and expressed in management debates has important implications for co-management, both from the perspective
of mobilising fishers’ knowledge for rational management and from the perspective of equitable control over the knowledge base.

Tacit knowledge is developed and used mainly through practical fishing activities. Pálsson [18, 19] examined skippers’ knowledge of fish. They can find it hard to explain how they know things. Fishing is learned mainly through imitation and Pálsson [19] found that this takes place often without discussion, or even consciousness. However, other researchers have found considerable variation in the degree to which fishing skills are passed on through discussion as opposed to imitation [20]. The way people learn skills is, in general [21], an important part of their identity and Roepstorff [22] found that “learning by doing” was an important part of fishers’ identities in Greenland. Possessing knowledge in a tacit form is not seen as a problem; on the contrary it is seen as an important professional and social asset. The question of tacit versus discursive knowledge has important implications for the knowledge base for fisheries management. Some of the fishers’ knowledge of nature, a part that may be linked to their identity as fishers, and hence of considerable political importance in management debates, is likely to be difficult to communicate either among themselves or with other stakeholders.

Discursive knowledge is knowledge that can be articulated and enables effective participation in and shaping of political discussions. ‘Discourses’—the sociological term are important for fisheries management because they are an important way to mobilise social power [23–25]. A large literature of discourse analyses has emerged in environmental studies, including fisheries. Charles [26] argues that the three basic discourses of conservation, rationalisation, and social/community explain most fisheries policy debates. Discourses link facts, values and interests in ways that reduce complexity and suggest particular outcomes, often in ways that emphasise particular facts over others. Discourses may become political symbols and take on a ritual character [27]. Palmer and Sinclair [28] argue that:

Appeals are sometimes made to incorporate local knowledge into fisheries management, without addressing the problem of exactly whose knowledge is to be considered most appropriate when there is disagreement. There is simply no single local vision or knowledge that is waiting to be heard—even within a single sector of the fishing fleet” (p 268).

As Davis [29] emphasises, it is just as important to look for variation in knowledge within groups as between groups, because the consequences of designating some groups as the embodiment of ‘knowledge’ have strong implications for the relative power of groups.

### 4.3.2 Oral vs. written knowledge

A second important distinction between the different forms of stakeholder knowledge is that between oral and written knowledge. While scientists and conservationists rely almost entirely on written information about the fishery, a considerable proportion of fishers’ knowledge is communicated in oral form. A good example, documented in Wilson and Degnbol [16, 17], is the idea among fishers in the United States that the East Coast bluefish stock had moved offshore in the mid-1990s. This was close to a consensus. Longliners who targeted swordfish reported that bluefish were stealing their bait much more than
in the past. Others said that they had heard the same thing from tilefish and wreckfish fishers, who fish in deep water canyons. One fisher’s observation was reinforced by that of another until a consensus emerged based on a great number of consistent observations. It is very likely, however, that any information that disagreed with the consensus would be dismissed as going against “common sense”.

Ong [30] summarises extensive research on the implications that oral versus written language has conveyed for the knowledge content. While he couches the distinction as between oral and literate cultures, a great many of his illustrations suggest that these comparisons can be made between oral and written fields, meaning arenas of social action [31], such as a professional ‘field’. Because oral knowledge relies on memory, the information takes on mnemonic, redundant and formulaic forms that aid the recall process. Oral information is additive rather than subordinative or analytic, meaning that it tends to be organised in flatter hierarchies than written information and involves fewer categories. Because oral information is not stored in a durable form, a much smaller amount of information is preserved and this store tends to be conservative in content, because effort is focused on remembering, rather than on adding new information. Oral information is less abstracted from the day-to-day tends to be related to the immediate and concrete. As such, although the oral versus written distinction is not the same as the tacit versus discursive distinction, we can think of oral information as being closer in this sense to tacit knowledge than is written information. Oral information also tends to be ‘agonistically’ framed, meaning that it takes it form in disputation, and is more defined in respect to oppositional views than is written information, which more often tries to ‘rise above’ disputes and be framed as objective. Oral communications are “empathetic and participatory” [30], rather than objectively distanced.

Furthermore, within the domain of written information there are also implications of the printed versus non-printed forms of knowledge. Beyond the simple fact that in printed forms information is widely reproducible, the printed form also leads to information being “finalised” [30] in a particular way, in the step between the creation of the text and its reproduction. In the publishing world, this step is actually highly ritualised. Oral information changes somewhat in nearly every telling, and even non-printed manuscripts are treated as drafts and subject to revision, but a printed manuscript stands on its own as a finished product. This has important implications for science; indeed, the invention of the printing press arguably made science in its current version possible because it made the reproduction of both exact observations and exact verbalisations possible [30]. This reproducibility also led to the important concept of the “black box”: the accepted discovery or procedure that is used to create more knowledge. What is and is not to be accepted as a black box is the focus of scientific disputes [32]. The development of electronic media, by making printed matter much more easily fungible, has important implications for this distinction between printed and non-printed forms of knowledge.

4.3.3 Anecdotal vs. systematic information

The replicability of printed knowledge also brings us to the last of the main distinctions between the forms of knowledge relevant to the knowledge base for fisheries
co-management, that between anecdotal and systematic information. This one is, of course, much more familiar in discussions of fisheries knowledge. It is a distinction, however, that applies to data—a set of individual observations—rather than to knowledge as such, which requires an understanding of the processes that link the observations. Systematic data is gathered, in principle, by procedures, which if they are not identical across time and space, then the way that they vary is known and can be accounted for. Systematic information is a way to link scale levels: it is a way to package information for transmission to a higher scale level to be used to characterise processes happening at that level.

To characterise information as anecdotal simply means that the observation it is based on was not made systematically and cannot be used to characterise phenomena at higher scale levels. It is not a reference to the reliability of the information, although the word is often used and understood that way. Replicable procedures of systematic information are expensive and specialised and are carried out only by resource rich stakeholders. This is usually governments; but, increasingly, certain elements of the fishing industry are able to generate systematic knowledge—under ITQ systems, for example. The replication of data gathering procedures across time and space is also an ideal that is difficult to meet in practice, because even governments have limited resources, and for many other reasons. Therefore, results based on these systems are highly vulnerable to attack by those stakeholders who do not have the resources to gather their own.

These different forms of knowledge are important because of the close link between knowledge and power. When it comes to participating in the give and take of participatory management, holding tacit, oral, or anecdotal knowledge rather than discursive, written, or systematic knowledge can mean real disadvantages. These disadvantages are not based on the knowledge being invalid nor on the unexamined assumptions of others about its validity, although such biases certainly play a part. They arise because discursive, written, and systematic information is easier to apply to the practical problems of complex, multiple-scale, multi-stakeholder co-management efforts.

4.4 THE KNOWLEDGE BASE AND WORLDVIEWS

Co-management’s knowledge base must reflect management objectives and be seen to be valid by all stakeholders. For this to be achieved, it is important for stakeholder groups to coordinate their actions. It is important to gather the local knowledge held by fishers and for user-groups to work collaboratively with governments to bridge gaps in knowledge, and in order that we may gain knowledge of fishers’ behaviour and how it impacts upon the resource. The choice of solutions reflects different ideas about the world. Policy objectives can be categorised in terms of three paradigms, worldviews or ideational systems: rationalisation, conservation and social/community [33].

The rationalisation paradigm focuses on wealth maximisation and economic efficiency goals, whereas conservation is based on long term fish stock protection and scientific assessment. In contrast, the social/community paradigm revolves around the belief that social and cultural institutions play a critical role in maintaining sustainable harvesting practices, including
habitat protection. Significant tension exists between these perceptions of what constitutes a sustainable fishery, particularly since each paradigm has different implications for ownership rights, responsibilities and the distribution of benefits (Charles 2001, cited in [11]).

In the USA and in Canada examples can be found of co-management regimes that contain a mix of different paradigms. In the USA:

The highly participatory processes have evolved from a regional and local scale of management decision-making that is accountable to the public trust and balances a range of ‘world views’, representing elements of each of the paradigms” [10].

In Canada, the situation is different:

The federal Fisheries Act gives the minister of Fisheries the exclusive and centralised authority for ocean fisheries, whereas in the USA this authority is shared between the federal and state governments at the regional scale and further tempered by various avenues for public participation [10].

However, in Canada, First Nations’ (indigenous peoples) collaboration with other coastal communities, aquatic resource interests, and federal and provincial governments, creates a forum where different ‘world views’ and paradigms can be balanced.

This is ideally what co-management should try to be: a mix of world views trying to find a balance between the needs, beliefs, rationalities and opinions of all participants. This ideal is, however, quite difficult to reach. Co-management requires a new paradigm: the democracy paradigm. Along with the sharing of responsibilities and power, knowledge must also be shared between scientific communities, government(s) and user communities. This balancing between and the mixing of different paradigms requires time and will be a struggle; it is a democratisation process. In the early 1980s, there was a struggle—between the Washington Department of Fisheries (WDF) and tribes that hired biologists and built their knowledge capacity—about tribal access to WDF’s stock abundance data, everyone’s harvest data, and the very definition of conservation [4].

The WDF did not want to reveal the paucity of its stock abundance data and the level of uncertainty surrounding its analysis and decision-making about the harvest. Furthermore it did not trust the tribes more than any other fishermen to report their catch accurately, especially since some tribes had asserted their treaty rights through illegal fishing for decades. (p 63)

This process took ten years and resulted in a complex power-sharing relationship in which the state and tribes agreed to work jointly on every aspect of data gathering, data analysis and harvest planning and eventually played complementary and mutually supportive roles [4].

Of course, ensuring all stakeholder groups are represented in a co-management initiative is difficult due to their large numbers [5].

At smaller scales greater communicative resources are available that […] make the steering of institutions more sensitive to the nuances of the many relevant social and ethical values.
and, just as importantly, are better able to identify and respond to good science and other kinds of factual truth [13].

Geographical communities also have some advantages in the shared meanings and common goals that they can draw upon to empower effective management [5].

4.4.1 Fishers’ knowledge

The integration of fishers’ knowledge and practices into modern management plans is one of the most attractive facets of co-management. Such knowledge in fisheries can be inter alia:

- where and when to fish to obtain maximum catches
- optimal/typical fishing routes
- annual/seasonal changes in local fishing conditions
- feeding and breeding patterns of fish
- social aspects of the community
- local geographical features which may influence fishing trips
- fish behaviour (shoaling/migrations)

However, fishers’ knowledge is too infrequently given systematic attention. Local knowledge is important to give a qualitative viewpoint in an uncertain system, and is likely to minimise negative social and environmental impacts of management and thus promote sustainability [34]. Furthermore, failing to include local knowledge in management plans constitutes a “gap” in the knowledge base. Historically, the knowledge base for fisheries management has been based only on formalised research, with no contribution from resource users. Fisheries co-management institutions are, thus far, (in most cases) limited to the implementation of regimes, and resource users have not been involved in the development of the knowledge base for management decisions. And yet, local knowledge is, after all, one of the main benefits of co-management programmes [34, 16].

4.4.2 Knowledge of behaviour

An important aspect of co-management is the knowledge about behaviour of fishermen—for effectiveness sake for the managers, but also for fishermen themselves to give them insight into the behaviour of their colleagues. Collective action only makes sense when it is known that other users (or most of the others) behave in the appropriate way. Fishermen need to communicate among themselves about their actual behaviour, with the aid of monitoring and enforcement mechanisms, with the important aim being to create transparency and trust. Fishermen need to know that their efforts towards sustainability (for example, restricting themselves) are not in vain. Some studies have found the majority of fishers (50–90%) complying with regulations [35]. However, non-compliance is often cited as the main cause of management failure in fisheries [35, 36]. There are a multitude
of factors that influence the compliance decisions of fishers, including the degree of consultation they have in the management process.

When individuals who have high discount rates\textsuperscript{1} and little mutual trust act independently, without the capacity to communicate, to enter into binding agreements, and to arrange for monitoring and enforcing mechanisms, they are not likely to choose jointly beneficial strategies unless such strategies happen to be their dominant strategies ([37] p 183).

The extent of enforcement of regulations greatly affects the level of commitment. Enforcement involves the monitoring and detection of violators and their prosecution and conviction. It is believed that fishers make their compliance decisions based on perceived probabilities of being caught breaking rules. However, next to this calculation fishers follow peer norms and values. In order to know if these norms are changing into improved compliance behaviour due to co-management, fishers need to know about the actual behaviour of their peers.

4.4.3 Shared marine resource knowledge

Fisheries management has to deal with all these different types of knowledge. However, the systematic, written and discursive types of knowledge are often chosen over the anecdotal, oral and tacit knowledge for practical and normative reasons. The marine resource is interpreted differently by formal researchers and resource users: researchers view fisheries in terms of “stocks” and issues are discussed and interpreted at the stock level; fishers, however, view the fish resource in terms of local abundance relevant to themselves. This difference in scales of knowledge is currently a critical problem in fisheries co-management [17]. Fishermen often complain that their views are not heard in decision-making fora. For co-management to encapsulate these cognitive features there must be a way of combating the problem of the segregation felt by fishing communities from the government, which relates to issues of the different scales of local knowledge and research-based knowledge [16].

A co-management system must develop mechanisms to reconcile formal scientific knowledge and local knowledge belonging to the fishing community, such that acceptance and validity is maintained. One approach may be to develop indicators of the status of the system [16] that reflect fishers’ knowledge and that are supported by the scientific community [38]. Indicators must then be: observable by the stakeholders; and within reasonable economic limits to allow sustainability, as understood by formal researchers and user groups relevant to management. This means that they should have associated reference points (limits, targets etc.) acceptable to the stakeholders and to the public.

\textsuperscript{1} The discount rates applied to future yields derived from a particular CPR may differ substantially across various types of appropriators. In a fishery, the discount rates of local fishers who live in nearby villages will differ from the discount rates of those who operate the larger trawlers, who may fish anywhere along a coastline. Local fishers hope that their children and children’s children can make a living in the same location. More mobile fishers, on the other hand, can go on to other fishing grounds when local fish are no longer available [37].
If these criteria for indicators are met, then it is possible to develop a shared knowledge base for co-management. However, finding indicators that match all these criteria may be difficult [17, 39].

It can be concluded that to incorporate different paradigms or worldviews into co-management regimes is a difficult and enduring democratisation process. This process will be a struggle between different groups and ideas that, through events and actions, will (ideally) slowly grow into a workable institution in which all involved stakeholders are willing to cooperate in the process.

Figure 4.1 outlines the process dynamics of co-management institutions and suggests a democracy paradigm for fisheries.

The three categories of phenomenon—‘group’, ‘actions, events, data and observations’ and ‘paradigm’—shown in Fig. 4.1 differ at the level of logic, but are related. Actions and events become meaningful within a paradigm, worldview or ideational system. This ideational system is sustained by a certain group in competition with ideational systems of other groups [40]. In a co-management system, there is more than one paradigm active, so, schematically, three ‘triangles’ can be drawn. Because stakeholders, like the scientific communities, fishers and governments, try to find solutions for a problem, these triangles overlap, but not qua paradigm, which is why Fig. 4.1 has three points.

Compositions of groups and actual events are concrete phenomenon that can be observed, while ideational systems and paradigms are fruits of human mind and imagination. Actual events and actions—such as changes in stock abundance, increasing oil
prices, new gear types, environmental disaster, new stock data, campaigns of environmental organisations, and political pressure—will be interpreted differently by the groups involved in the process. These interpretations of actual events and actions give insights in the ideational system of the groups.

For ideational systems or paradigms, several features are important, according to Schuyt [40]:

• Every ideational system has its own (linguistic) way of expression.
• It is an abstraction of reality.
• Ideational systems coexist with other paradigms; there is a plurality of perspectives in society.
• Paradigms may change, although not randomly because change is related to actions and events and group composition. When the composition of a group change this will also influence their worldview and, consequentially, the meaning they give to actual events and actions. Equally, when new events occur, they influence ideational systems. (This dynamic is expressed in the triangle by the arrows.)
• People have the capacity to put themselves in another’s perspective.
• Ideational systems clash with each other, therefore tolerance for other perspectives needs to be developed.
• Paradigms ignore certain important aspects of reality. There seems to be a systematic neglect of aspects that, in that moment, for that type of problem and type of acting, are judged as unimportant. When claims of the only truth are made, dogmas and professional blindness arise.
• It structures reality, and, in that way, forms a guiding principle for acting.
• Ideational systems are partly anchored in deep nearly unconscious metaphors.
• When one ideational system is seen as the only right system, one speaks of ‘preoccupation’. Experience and reality can then only be seen through the glasses of this dominant perspective.

As co-management implies a balance between different paradigms, tolerance and respect for different perspectives needs to be developed. Acceptance of the fact that one’s own perspective contains blind spots that can be filled through dialogue with representatives of other paradigms is part of the institutionalisation process of co-management. Groups and their worldviews have the inclination to struggle for dominance; thus, norms are needed to regulate the balance between these worldviews. This process is an internally and externally influenced dynamic process, and asks for a flexible co-management institution that creates satisfying fisheries management solutions through the democracy paradigm. In theory, this democracy paradigm contains all the perspectives, or certain aspects of these paradigms. By internal events and actions, such as meetings, workshops and other communications, scientists and fishers (and other stakeholders) can collaborate together in research to create management solutions that reflect particular and common concerns. In the course of time a common knowledge base for co-management can be developed in this way.

Below, some models and examples of co-management-like collaboration in research are presented.
4.5 MODELS OF ACTION TOWARDS DEVELOPING A KNOWLEDGE BASE FOR CO-MANAGEMENT

Research collaboration between fishers and scientists has a long history, dating back to the beginnings of fisheries science. Several collaborative research initiatives are currently taking place in Europe. One of the most active initiatives has been the North Sea Commission Fisheries Partnership (NSCFP), which was organised by the North Sea Commission—an organisation whose members are governments of sub-national regions around the North Sea. Their fisheries interest group took the lead in setting up a collaborative process between fishers and the International Council for the Exploration of the Seas (ICES, see Chapter 8). The NSCFP has led to regular, informal reviews of ICES advice by scientists employed by the fishing industry and has also been instrumental in creating the North Sea Regional Advisory Council (or RAC). In 2002, ICES and the NSCFP created a Study Group on the Incorporation of Additional Information from the Fishing Industry into Fish Stock Assessment (SGFI) that met annually from 2002 to 2005 [41].

One way of understanding these research collaborations is to suggest that there are four basic models of collaboration [42]. These models are cumulative in that each one incorporates the basic perspective of the earlier ones. *The Deference Model*: Scientists are the experts and the best way to get an accurate picture of nature is to rely on their professional judgement. *Experience-based versus Research-based Knowledge*: Scientists and fishers see the world differently because of differences in training, experience and culture. Science can be improved by listening to what fishers have to say about the resource. *Competing Constructions*: Scientists and fishers collaborate within interest groups to create pictures of the resource that reflect particular concerns. *Community science*: Collaborative fisheries science conducted in the context of cooperative management.

4.5.1 The deference model

The deference model is the most widely accepted “common sense” idea of science as a social process. Scientists are the people that society has trained and given the institutional and physical tools to decide what is true about our natural environment. That is their job and they are the best ones to do it. The most widespread of this type of collaborative work is data gathering for scientists, in which fishers and others act as research assistants: a very common example is tagging studies. Other common activities include fishers participating in research through providing research platforms, logistical support and at-sea collaboration. Sampling efforts have been improved in both Belgium and Norway through the creation of “reference fleets” that provide more detailed information about catches and landings than are generally required by sampling programmes and, therefore, help make improvement in the sampling programme methodologies. In Germany, scientists have been sampling on board fishing vessels for 15 years or more. Programmes for monitoring discards through voluntary observer programmes were pioneered in Scotland and are now

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\[2\] Much of the material in this section was presented earlier in Wilson [42].
widely found in Europe. Fishers volunteering to provide information from their logbooks, which is more detailed than the information that is generally available, are also common. Observer programmes also include monitoring of bird and marine mammal bycatch rates [41].

4.5.2 The experience-based knowledge model

The experience-based knowledge (EBK) model both incorporates, and is a challenge to, the deference model. No one claims that EBK includes generalisable information. The emphasis is on finding local information that can supplement research-based knowledge. In this sense, the EBK model builds upon the deference model. However, the EBK model makes two important new claims. The first is that EBK, which reflects the thinking of fishers, is critical to producing a picture of nature that will be accepted as the basis of management. The second is that EBK, while not generalisable, is just as valid in a local context as RBK. Thus, the scientist is no longer assigned the role of the gatekeeper who has the final word on what is or is not known to be true about nature.

One type of collaborative research within this model found in Europe is research on fishing behaviour. In Denmark, for example, such research has focused on changes in behaviour caused by the implementation of technical measures. Another very common type is research into the development of new fishing gear. A great deal of work has been implemented in both Europe and North America on the development of devices to prevent various kinds of bycatch and to increase gear selectivity. Under the “Science/Industry Partnership”, Scottish fishing vessels were chartered for both gear trials and biological data collection programmes. Researchers at the University of Aberdeen and the Fisheries Research Service (FRS) in Scotland have been interviewing skippers to help understand how fishers make fishing decisions, to assess their views on the state of the stocks, and general issues of fisheries biology and ecology. Fisheries scientists from England, Scotland and Germany meet with representatives of the fishing industry before and after ICES assessment working groups to learn fishers’ views on the state of the stocks, the strength of current year-classes, and changes in fishing effort [41]. The North Sea Fisheries Partnership, in cooperation with the North Atlantic Fisheries College, conducts an annual survey of fishers’ views of the trends in the abundance of major commercial stocks.

4.5.3 The competing constructions model

Scientists are increasingly working with fishers, and other interest groups within fisheries, in pursuit of particular policy objectives. People select different facts from fisheries science to put together an overall picture of the resource that fits their needs. In spite of the insults about “disinformation” that get bandied about when tempers flare over management disagreements, there is not usually anything sinister about this. The scientific questions that people ask, and the answers that they find useful, are always related to some agenda. This is true in the highest scientific “ivory tower”; and it is certainly true throughout fisheries science. Asking different questions and using facts in different ways builds different pictures of the resource. This, however, does not mean that there is no
such thing as an objective fact and it does not mean that all descriptions of the resource are equally valid.

Because they are confronted with different questions and problems, each of the major players in fisheries management, and the scientists that work with them, tend to construct a version of nature that fits their needs. This means, to put things as simply as possible:

(a) scientists working in, or in the service of, management agencies tend to construct a picture of nature that is more amenable to bureaucratic management than it really is;
(b) environmentalists, who always have to solve the problem of mobilising their constituents, tend to construct a picture of nature that is more threatened than it really is; and,
(c) fishers tend to construct a picture of nature than can sustain more fishing than it really can.

Honest collaborations often happen between scientists and fishers working toward particular policy objectives. In most cases, these groups are convinced that their case has merit and are prepared to do what it takes to convince others of their point of view. This means selecting the most defendable scientific facts that suited their goals. This model of collaboration between fishers (and other interested parties) and scientists is perhaps the most common kind of collaboration of all. It happens every day wherever a group or agency that has an interest in fisheries employs scientists. Each of the major fishing industry organisations in Europe has fisheries scientists in their employ, as do the larger of the conservation NGOs. These scientists actively represent the interests of their groups in policy discussions. More and more fora are emerging where scientists playing advocacy roles are able to come together and represent their views and look for possible compromises. In England, the Fisheries Conservation Group includes representatives of the fishing industry and the Centre for Environment Fisheries and Aquatic Sciences (CEFAS) Laboratory who meet several times a year to discuss fisheries management issues, with respect to both national and EU policy negotiations [41]. ICES has begun to experiment with allowing representatives of various industry and conservation groups to observe the meetings of the Advisory Committee on Fishery Management (ACFM), where ICES fisheries advice to the European Commission is finalised. The RACs are the latest and most advanced experiment in bringing together different scientific perspectives on management issues (Chapter 9).

A “devolution movement” occurred in the New Zealand commercial fisheries in the mid to late 1990s. This was formally recognised in 1999 when the Fisheries Amendment Act delegated certain responsibilities to stakeholder groups, referred to as commercial stakeholder groups (CSOs), essentially composed of ITQ owners who take control of managing the resource [43]. Fisheries organisations hire their own experts to bridge the gap between scientific knowledge and user group knowledge. In the last 25 years the fisheries management system in New Zealand has been reformed continuously. The New Zealand co-management system is of particular interest because it has only recently been developed and, due to the difference in its fishing industry compared to many other countries (the relatively new expansion of the deep water fishery and the lack of strong community-based regulations, apart from the Maori), it is somewhat different to other
types of co-management outlined in the literature [43]. Management of the New Zealand rock lobster fishery has been based on a certain degree of cooperation at the local and national levels since the 1980s. The introduction of the ITQ system in New Zealand provided further incentives to enhance cooperation of regional and national organisations and the government to manage the rock lobster fisheries. Development of co-management in New Zealand resulted from a combination of industry activity (at the local and national levels) and from strengthening property rights through the implementation of ITQs. As discussed below, there is some evidence that these efforts are leading to a shift from purely competing constructions to the fourth model of collaboration, which we term community science.

4.5.4 Community science

Many people involved in fisheries are creating innovative institutions to bring about collaborations in science that recognise that these differences are going to exist and use open communications to try to move beyond them. They are bringing the dynamics of “community” into the fisheries science process in the sense that encouraging open communication both increases understanding and makes management institutions more sensitive to new developments in the ecosystem, hence facilitating adaptive management [13]. There are many examples of these efforts. One of the oldest examples is the Fishermen and Scientist Research Society, discussed below, which began in 1993 in Nova Scotia.

These collaborative efforts incorporate the other three models. They defer to the expertise of trained scientists and give them the leadership roles in the collaborative process. This expertise and leadership role, however, is not based so much on the assumption that the scientists should be the gate keepers of what is true about nature, but rather on the scientists’ ability to present their understanding of nature in a transparent way, to explain rigorously how they know what they know. This kind of collaboration also respects the importance of EBK and takes into consideration that fact that it can be very difficult to put EBK in a form that is easily understood by other stakeholders and which fits easily into management decision-making. The collaborative efforts take into account the fact that different understandings of the resource, and different ideas of what should be done, are going to reflect the practical problems in people’s fishery-related lives. It is noteworthy that most of these programmes emerged in areas with serious resource depletion problems, and that some of them were a response to intense conflict between stakeholders.

A number of examples of these kinds of collaborations are emerging in Europe. In England, the industry group NFFO and the CEFAS laboratory now have an established network of people from principal fishing areas to facilitate direct access and dialogue. This network has been particularly active in the development of stock recovery programmes [41]. A similar network has been developed in Germany over the last 10 years. One German fisheries scientists reports that:

Over the years a quasi symbiotic relationship has developed between some of the researchers and skippers. The scientists gain a lot by listening to the fishers and regularly bring home
very important information about the behaviour of the fleet, the abundance of the stocks and
the actual fishing situation. Some of the fishermen on the other hand benefit from discussions
with the scientists and learn about fishery biology and what the data collected are used for
[41].

In New Zealand, the NZ RLIC is an umbrella organisation composed of nine regional
stakeholder organisations, referred to as CRAMACs [44]. The CRAMACs are comprised
of quota owners, processors, exporters and fishers [45]. Each CRAMAC elects a repre-
sentative to the board of the NZ RLIC where the national total allowable commercial
catch is decided. The NZ RLIC and the CRAMACS share a mutual bottom–up/top–down
relationship, where information, expertise and responsibilities flow in both directions [44].

Collaborative research programmes are commonplace in New Zealand, which is an
indirect outcome of their rights-based fisheries management systems. This outcome came
about in response to a combination of fisheries management institutions and processes
that (a) create incentives for fishers to take more responsibility for fisheries research; and
(b) to improve the transparency and legitimacy of industry-led research. New Zealand
spends approximately 2.5% of the value of landings on improving the state of fisheries
knowledge, with the primary aim being the sustainability of New Zealand fisheries [45].
The NZ RLIC is actively involved in the multi-stakeholder organisation called NRLMG
(National Rock Lobster Management Group), which is made up of governmental and
environmental non-governmental representatives, fishers (commercial, recreational, and
indigenous) and scientific advisors.

The NRLMG is recognised as the primary source of information provided to the New
Zealand Government, and, as such, the NZ RLIC has become an accredited provider of
knowledge to the Minister of Fisheries. The NZ RLIC undertakes its research programmes
in collaboration with scientists and stock assessment consultants. The NZ RLIC, now one
of many such organisations in New Zealand’s fisheries, paved the way for collaborative
research and for developing the knowledge base of fisheries management. They view the
local knowledge that fishers possess as essential to successful fisheries management, and
believe that the role of fishers in collaborative research should not just be considered as
a source of funding or an additional source of scientific knowledge [45].

In Ireland, a co-management programme for pelagic fisheries has been established that
is empowered to invoke both voluntary measures and national by-laws. While this is pri-
marily a co-management programme, it has also developed an active research programme
that includes fishers recommending areas for additional research, recommendations that
are backed up financially by both the fishermen’s organisations and the state [41]. Another
cooperative research project is the F-project in the Netherlands. This project began through
what was essentially a deference model with an intent to develop a reference fleet. The
project ran into considerable problems with fishers withdrawing from the programme
both because of the way it was structured and because of a general distrust of the man-
agement system. However, maintaining close contact with participating fishers, including
a feedback system, was seen as critical from the very beginning of the project, and pro-
viding feedback is seen as an important and integral part of the study. As a result, the
problems began to be reversed when fishers took a more directive role in the project,
including in one instance a computer-literate fisher being the one who set up the database
Hoefnagel et al. programme for the project ([46]; Wim van Densen, personal communication, August 1999). As a consequence of this improved contact between fishermen and biologists, Dutch fishermen have now monitored their plaice discards every Tuesday and Thursday around 16:00 since the beginning of 2005. Since data on discards were previously mainly based on estimations, a more reliable database can now be built (Nathalie Steins, personal communication, 9 January 2006.).

An Asian example of community science can be drawn from the Philippines. The Philippines has the largest number of community-based resource management (CBCRM) programmes in the world [47, 48]. The rationale of CBCRM projects is to put the emphasis on the increased participation of fishers in fisheries management [47, 49]. The reason CBCRM projects have been so successful in the Philippines is that such projects hold within them a vast amount of local knowledge for their successful implementation [50]. The Philippines has over two decades of experience with coastal resource management, in particular, with community-based management regimes, from which other Asian countries are drawing information [51]. Over a ten-year period, from 1984 to 1994, over 100 CBCRM projects were initiated in all areas of the Philippines [50]. Most of these projects were supported by local governments [49], and their numbers are growing at a phenomenal rate [50]. A case in point is Cotong Bay, where local communities have managed their coastal resources since the 1940s. During the 1960s and 1970s, the mangrove resource of Cotong Bay became heavily exploited and, as a result, catch rates per trip steadily declined. Consequently, between 1989 and 1991, a government supported co-management scheme was introduced; and the local community, external groups, and local and national governments collaborated to establish the state of the situation and to develop plans to allow the recovery of the mangrove resource. Through the development and improvement of knowledge of the resource, and from the establishment of responsibility and accountability, there was a shift from “resource use” to “resource stewardship”, and there are now signs of resource rehabilitation [49].

In North America3 two types of collaborative projects fit the science as community model. The first is programmes initiated by the fishers themselves. The Fisheries Survival Fund (FSF) is a group of seventy plus scallop boats based mainly in New Bedford MA. This group initiated a collaborative research project that includes the National Marine Fisheries Service, the University of Massachusetts at Dartmouth, and the Virginia Institute of Marine Sciences. This coalition looked at scallop abundance in areas off the coast of New England that had been closed as a way to protect the groundfish stocks. The data they gathered convinced the government to open a portion of the area to scalloping. The research effort continues (Jim Kendall, personal communication, August 1999). The FSF, in a way, straddles the competing constructions and the science as community models. While they are definitely an ongoing multi-stakeholder coalition, some environmental groups have criticised the research and its aims.

The Southeast Alaska Regional Dive Fisheries Association (SARDFA) unites 560 fishers from five communities who are diving for sea urchins, geoducks and sea cucumbers. Their basic mandate is community economic development and their philosophy is that

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3 These descriptions of North American examples were presented earlier in Wilson [42].
management should go hand in hand with the development of a fishery, rather than waiting for a crisis. Dive fishers and processors have been supporting research and management with their own funds in order to supplement the work of the Alaska Department of Fish and Game. This collaboration occurred on an ad hoc basis until the creation of SARDFA, a process which began with local government support and culminated in its official creation through State legislation (Julie Decker, personal communication, August 1999). Julie Decker, the Executive Director reports that ongoing research collaboration with the state has led fishers to be more cohesive and organised because they are forced to work out clear proposals for research. They are beginning to learn to participate in research design and the interpretation of the results. She reports a slow, ongoing increase in trust between the fishers and the scientists that is creating a more responsive management system, but one that requires more effort to achieve (Julie Decker, personal communication, August 1999).

While SARDFA represents an effort to avoid crisis and conflict, the Fundy Fixed Gear Council (FFGC) emerged from crisis and conflict. Fisheries in South-west Nova Scotia experienced a series of protests in 1996 due to some controversial government management decisions involving ITQS, quota allocations, and other issues. The protests led to a mediation through which the fixed gear quota was divided geographically and this gave an opportunity for a co-management effort. The FFGC was set up to manage the fixed gear in three counties. They have organised their science activities around a Research and Advisory Committee that includes fishers, community members, environmentalists, community development workers and scientists. The FFGC emphasises an ecosystem approach to their collaborative work on stock assessments, habitat mapping and other research goals [52].

One of the most established of the cooperative research groups is also in Nova Scotia: this being the Fishermen and Scientist Research Society (FSRS), which began in 1993. It is a partnership between 156 fishers and 42 scientists that emphasises participating in and enhancing stock assessments by providing “information that only fishermen can obtain on a daily basis” [53]. The FSRS does not lobby. It sets its own research priorities, but it also supports itself through specific research projects conducted on a contract basis with DFO and other agencies. King [53] reports that building trust between the fishers and the scientists has been both their greatest challenge and accomplishment. The key to success has been timely feedback and direct communications.

The FSRS began with a pilot programme involving both fishermen and scientists; the programmes detailed below were not initiated by groups made up of fishers.

One group with a particularly interesting start began when an environmental group, the Conservation Law Foundation, enlisted Dee Hock, the founder and CEO of Visa and Visa International, to find ways for the different groups in the Gulf of Maine work together. The result was the Northwest Atlantic Marine Alliance (NAMA). They are helping to set up meetings to identify research priorities and establish protocols (Craig Pendelton, personal communication, August 1999). Craig Pendelton, a fisherman and NAMA’s Coordinating Director, says that groups are planning research together that have never communicated before and conducting the fisheries science work is part of the “glue” that keeps them together. It is still a struggle, however, to encourage the industry to participate at “a more
professional level” and influencing management still feels like “taking down the iron curtain” (Craig Pendelton, personal communication, August 1999).

The major impetus for the St. Georges Bay Ecosystem Project has come from Xavier University and DFO. The programme’s partners include native and non-native fishers’ organisations. The programme takes an ecosystem-based approach to management and is involved in collating as much data as they can about St. Georges Bay, including both the results of mainstream science and EBK. The goals of the programme also include improving the working relationship between harvesters and scientists and developing the harvesters’ scientific literacy. This project is in the area affected by recent, intense conflicts over native fishing rights, however, and this will certainly present the project with some overwhelming challenges in trying to meet these goals. Hopefully, it will be in a position to make a contribution to moving the community through these problems.

These types of cooperative programmes are also found in freshwater, recreational fisheries. Freshwater fisheries management in Nova Scotia involves considerable open interaction between the Inland Fisheries Division of the Department of Fisheries and Aquaculture (DFA) and the recreational fishing community, which has a number of opportunities to raise concerns about both existing management and the need for more management (Michael Robinson, personal communication, August 1999). They work through provincial angling organisations that are deeply involved in collaborative research and often take responsibility for major research programmes. The Nova Scotia Salmon Association heads the ‘Adopt-a-Stream’ programme and the River Watch program, and The Fisheries Institute of Nova Scotia takes the lead in examining proposed legislation that may affect the fisheries. Extensive special research efforts are coordinated by angler associations. For example, the Canadian Association of Smallmouth Anglers (CASA) has just completed a five-year study of Smallmouth bass in conjunction with Inland Fisheries—the parameters of which were negotiated between CASA and scientists from the DFA (Michael Robinson, personal communication, August 1999).

4.6 CONCLUSION

The challenge for a co-management regime is to incorporate both resource users’ knowledge and scientific knowledge. In order to meet this challenge, the innovative institutions to bring about collaboration between communities of users and scientists must be created. The experiences described in the last section suggest that the dynamics of “community” can be brought into the fisheries science process in the sense that encouraging open communication both increases understanding and makes management institutions more sensitive to new developments in the ecosystem, thus facilitating adaptive management.

This institutional reform, like the development of a co-management regime, must be regarded within a larger societal and institutional context because it is often in this larger context that opportunities for major change in fisheries governance emerge. The socio-political settings of countries affect the type of co-management regime to be put in place, allowing and excluding certain arrangements. For example, societies not familiar with political empowerment may find it difficult to share responsibilities with government, and, vice versa, government may find it difficult to devolve management tasks.
Human and financial capacity-building in communities will be a pre-condition for successful co-management institutions. Highly participatory processes should evolve from a regional and local scale of management decision-making, which is accountable and balances a range of ‘world views’, representing elements of each of the rationalisation, conservation and social/community paradigms. Co-management should aspire to be a mix of world views, trying to balance between all needs, beliefs, rationalities, opinions, and knowledge of all participants. Co-management requires a new paradigm: the democracy paradigm. Next to sharing responsibilities and power, knowledge must be shared between scientific communities, government(s) and users’ communities. These communities have their own world views that give a certain perspective or rationality to their actions. By assuming a pluriformity of rationalities within a policy process, a new more complete perspective on management problems will emerge. Different forms of research collaboration between fishers and scientists exist nowadays.

The Community Science model is the (never actually achieved) ideal model for actions aimed at developing the knowledge base for co-management. It is based on the belief that scientists and resource users see the world differently due to their different experiences, and, as such, is used to support scientific findings. In this model, the scientist is no longer “in charge” of what is known about the research; the point of view of the scientist is combined with the points of view of other users. They draw on the knowledge of science and respect experience-based knowledge (EBK) and take into account the competing constructions of stakeholder groups to resolve conflicts of opinion. The creation of these new institutions, which are, to varying degrees, already in operation in, for instance, New Zealand, the USA and Canada, is a democratisation process and asks, consequently, for open mindedness from all participants.

REFERENCES

Knowledge base of co-management


5.1 THE THEORY OF TAXES AND SUBSIDIES

5.1.1 Market failures and taxes

Financial instruments in terms of taxes and subsidies have often been used to protect one country’s industry against competition from other countries’ industries, or to correct market failures including low productivity and income development in certain sectors compared to others. Further, financial instruments have been used to improve the working environment and the safety conditions of exposed population groups and branches. The use of financial instruments in trade entails distortions and welfare losses, and is considered harmful as it leads to an increase in inefficient domestic production, at the expense of a decrease in the supply shares of efficient foreign production. These types of financial instruments are the main subjects of the negotiations in the World Trade Organisation (WTO). If the fishing industry is protected in the same way by the use of financial instruments this will lead to further overexploitation of the fish stocks (lower supply) of the domestic country, and less overexploitation (higher supply) in the exporting countries. This is a result of the biological characteristics of natural resources, such as fish stocks, which cause the supply curve to bend backwards with increasing prices, compared to almost all other types of production, where supply increases with increasing prices.

The use of financial instruments in the fishing industry could be justified, however, as the industry is characterised by market failures. These failures are caused by external effects because the harvest of one fisherman directly impacts the harvest opportunities of other fishermen [1]. The result is over-fishing, which means that the harvest could actually be increased if effort is reduced. These types of externalities were recognised as early as in the beginning of the Twentieth Century by the economists Jens Warming in 1911 [2] for fisheries and A.C. Pigou [3] for external effects in general. According to this theory, these market failures can be corrected by the use of correcting taxes (often named Pigou taxes). Within the WTO, these types of market distortions were not high on the agenda until the beginning of the twenty-first Century.
When financial instruments are used to correct market failures, two effects are recognised: the efficiency effect—optimal resource allocation; and the distributional effect—which groups will win, and which will lose. The effect of financial instruments, in terms of taxes, on a fishery is shown in Fig. 5.1. Fishing effort (or fishing mortality) is measured as an index on the horizontal axis. The vertical axis measures the catches and the total costs and revenue from the exploitation of the fish stock. Catches, gross revenue and total costs are shown as functions of fishing effort where gross revenue is calculated from catches multiplied by demand prices. In a fishery that is not managed properly, the fishing effort will be driven to the intersection between total costs and gross revenue. This would entail lower catches than possible and overfished stocks.

In a fishery managed by use of a tax, fishing effort and overfishing are driven downwards as the use of a tax will increase the total costs of fishing from the fisherman’s point of view; although not from society’s point of view that considers taxes as transfers. In principle, reduction of effort and overfishing could be achieved not only by use of taxes, but also by restricting effort, by use of subsidies, or by use of property rights that induce the fishermen to take the external impact directly into account. In the following, only taxes and subsidies will be commented on with respect to efficiency and distribution of income. The effect in terms of optimal fishing effort of using either measure could be the same but the distributional effects are different depending on the type of measure. It very much depends on the design of the financial scheme, and that fishermen and society are impacted differently by the measures.

From a socio-economic efficiency point of view, taxes will reduce fishing effort, and reduce costs of fishing, preserve the fish stocks at a higher level, and increase landings.

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1 Growth and fishing mortality rates are derived from plaice in the North Sea and imputed into a cohort (age-structured) model.
2 Subject to a price flexibility at -0.5 (price elasticity at -2).
Subsidies will, in general, lead to the opposite, if the fisherman uses the subsidy to reduce his costs of fishing (cf. paragraph 5.1.3). The effect of a subsidy is lower costs leading to higher fishing effort and lower catches, gross revenue and fish stock abundance.

However, there are three parties to consider in this scenario: the fishermen that must leave the fishery; the fishermen that will stay; and society. If a tax is imposed, society will collect tax revenue, as indicated by the vertical tax-arrow in Figure 5.1. The fishermen that stay (producing fishing effort from 0–0.175) will not be better or worse off but they will have to pay a relatively high tax. The fishermen who have to leave the fishery will save costs, but they will also be deprived of earnings. These fishermen could, however, be compensated from some of the tax revenue.

In practice, it is very difficult to estimate optimal fishing effort, optimal taxes, and redistribution of taxes: see Clark [4–8]. Further, lack of information and hidden incentives of the fishermen will play an important role for the outcome of the intervention: see Frost et al. [9]; Jensen and Vestergaard [10]. It is obvious, however, that it is possible to move from a worse situation to a better situation, although not necessarily the best. Yet, in general, it cannot be argued that a tax scheme will be better than a subsidy scheme. It depends on the design, as the choice includes not only efficiency gains but also how these could be achieved with respect to priorities of income distribution and the acceptance of the industry.

To operate and assess consequences of taxes and subsidies a knowledge base requires objectives and information about the type of tax or subsidy. The WTO operates with green, amber, and red types where green are acceptable, red are harmful, and amber are in between.

In this chapter, taxes and subsidies that impact trade are, to a large extent, disregarded. Instead, the focus falls on financial instruments that have environmental effects—for example, those that impact on fish stocks. In order to examine the consequences of such instruments, it is first necessary to obtain information about the functioning of bio-economic systems. Or, in other words, to assess the different consequences of such taxes and subsidies a knowledge base is required, detailing both the objectives of, and information about, the type of the tax or subsidy. With regard to the first, the determination of management objectives is essential for the successful operation of a tax or subsidy system relating to the exploitation of fish stocks. Frequently and currently, management objectives are fixed with respect to the abundance of fish stocks or catches, with the aim being to prevent overfishing. These objectives are insufficient as financial transfers are economic in nature. Therefore, an outline of a bio-economic system is required as a framework in which to place the biological and economic information that is essential for managing fishing by the use of financial transfers. Second, a clear understanding of different tax and subsidy instruments is necessary for managers to decide how impacts could be assessed.

### 5.1.2 Definition and types of financial instruments

The use of financial instruments, or ‘financial transfers’, impacts on the profit from fishing: either by changing the price of fish obtained by the fishermen; or by changing the costs of fishing. For example, financial instruments such as taxes and subsidies imposed
on prices and costs have a direct impact on the fishermen’s profit. Meanwhile, taxes and subsidies imposed as ‘lump sum’ payments influence the opportunities of the fishermen in the sense that, for example, accepting a subsidy and leaving the fishery would make them better off economically than if they stayed.

One definition of financial transfer is: “Government action (or inaction) that modifies (by increasing or decreasing) the potential profits earned by the firm in the short, medium and long term” [11].

MacAlister Elliott [12] considers that this definition is deficient in one area: it fails to define which government action is legitimate, and should not be considered a subsidy [13]. The term ‘financial transfer’ covers both charges (taxes) and subsidies. Nevertheless, practical examples of their implementation are strongly biased towards subsidies since that form of their use is much more extensive. Table 5.1 details the range of possible subsidies that are applicable to fisheries management.

A ‘charge structure’ includes charges under different names such as taxes, charges, duties, tariffs, fees and even fines. However, the different names often relate to specific

<table>
<thead>
<tr>
<th>Subsidy set</th>
<th>Definition</th>
<th>Examples</th>
</tr>
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<tbody>
<tr>
<td>1 Direct financial transfers</td>
<td>Government financial transfers that reduce the costs and/or increase the revenues of producers in the short term</td>
<td>Direct payments by government to or on behalf of producers, e.g., grants to purchase or modernise vessels and income support payments</td>
</tr>
<tr>
<td>2 Services and indirect financial transfers</td>
<td>Government interventions—regardless of whether or not they involve financial transfers—that reduces the costs and/or increase the revenues of producers in the short term</td>
<td>Tax waivers and deferrals, as well as insurance, loans and loan guarantee provided by government. Government provision of goods and services below market prices. Closely correspond to many WTO definitions</td>
</tr>
<tr>
<td>3 Regulations</td>
<td>Set 2 subsidies plus the short term benefits to producers that result from the absence or lack of interventions by government to correct distortions (imperfections) in production and markets, which can potentially affect fishery resources and trade</td>
<td>Implicit benefits to producers that are associated with a lack of government regulations requiring producers to bear the costs that they impose on other parties (e.g., environment, such as catches of sea turtles and marine mammals). Another example is where government does not do enough to prevent the overexploitation of a fishery resource, which in the long term imposes costs on others and themselves</td>
</tr>
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</table>
services or products. The conceptual difference between these items (names) is that a tax is a payment to the public sector without any direct service in return, whereas charges, duties, tariffs, and fees are payments that are linked to the provision of direct services and products. Fines are used if rules are violated, for example if fishing quotas are exceeded.

The use of a ‘charge structure’ in fisheries is based on the premise that fish stocks represent a national resource and that society, as a whole, should receive an inter-temporal share of the benefit flow of their sustainable exploitation. As a result, the markets failure of the fishing industry is fully or partly corrected. Following Hatcher and Pascoe’s [16] classification different charges include:

Access or entry charges are fees that are collected simply for access to the fishery (for example, an annual licence fee) and, in theory, take no account of the actual amount of use that is made of the resource (although in some cases some weights on landings, vessel type, and so on, can be introduced).

Use charges are variable charges, which depend on the amount of use that is made of the resource. Use may be defined either directly, in terms of output (landing taxes that reduce revenues to all fishermen in the same proportion), or indirectly, in terms of inputs (input taxes in one or more components of effort, thereby increasing variable costs). Landing taxes do not have a distortional effect but create enforcement problems; while effort taxes induce input substitution, which results in sub-optimal combinations of inputs.

Cost recovery charges are levied on fishermen to recover the management costs. They are founded on the theory that the main beneficiary of management, the harvesting sector, should bear at least a part of the costs of providing these services.
Rent capture charges. Many countries base their fishery management system on the principle that fisheries are a community owned resource, even if access rights to a fishery can be privately owned. When tradeable fishing rights are introduced in a fishery and given for free to those fishermen already participating in it (not sold off by public auction), these first round owners get the resource rent capitalised into the market value of those rights when they sell them, unless the management authority impose a charge to provide a return to the community from the use of marine resources. These charges are set over and above the costs of management.

In what follows, the term ‘charges’ is used as the general term for payments from the industry (and the consumers) to the authorities, and the term “subsidies” is used for payments from the authorities to the industry (and the consumers). This distinction is clear, although, other specific terms within these two groups will be used from time to time. In particular, the word ‘tax’ is used in cases where the payment is not linked to the provision of services or products.

5.1.3 Impact of subsidies

Over the past decade there has been increasing attention given to the perceived impacts of financial interventions in fisheries [17–22]. Not only is there the potential of financial instruments to distort markets and trade patterns, but there is also the possibility that they provide fishermen with increased incentives to fish already fully exploited or overexploited stocks, which has implications for long term sustainability [23]. The views of international organisations with regard to the impacts of various types of charges and subsidies are discussed below.

It appears from Fig. 5.2 that subsidies affecting demand (gross revenue) and production costs directly or indirectly are harmful, in the sense that they lead to further overexploitation of the fish stocks and welfare loss to society. Further, if subsidies are used unilaterally they cause competitive distortions: see PricewaterhouseCoopers [24, chapter 8].

If subsidies are designed properly to incite fishermen to leave the industry rather than to increase effort, a subsidy larger than the difference between production costs and gross revenue of the fisherman will make him better off leaving than staying in the fishery. The first units of effort to be decommissioned (starting from effort index 0.9 in Fig. 5.2) will be relatively cheap, but the subsequent units will be increasingly more expensive, as the distance between the gross revenue per vessel and the production costs per vessel increases. Given that it is possible to determine the optimal decommissioning grants, the result in terms of fishing effort will be the same in the end as with a tax. However, by the use of decommissioning grants, society will incur an expense and the fishermen who stay in the fishery will experience a substantial increase in earnings equal to the tax revenue. The fishermen that have to leave will be compensated. Therefore, the effect in

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3 Resource rent is the maximum economic surplus that could be extracted from a fishery while the fishing industry continues to operate efficiently.
terms of optimal fishing effort will be the same although the distributional effects will be quite different between the three parties.

A further understanding of the way subsidies impact supply and demand in fisheries can be gained by the use of a conventional demand–supply exposition of the neoclassical economic theory. The supply of the fishing industry is characterised by backward bending curves that reflect the market failure in fisheries. Figure 5.3 below is derived from the same empirical data as Figs. 5.1 and 5.2, and it shows directly the intersection between demand and supply. From Fig. 5.3 it appears that if demand increases as a result of, for example, price subsidies (demand curve moves upwards) the supply will decrease, which is contrary to almost all other businesses. For natural biological reasons the supply is constrained by the maximum yield a fish stock can produce, which determines the point at which the supply bends backwards.
Because of the market failures in fisheries, certain subsidies are not harmful and could be characterised as ‘green subsidies’ [25]. Such subsidies could take the form of decommissioning grants if they are designed correctly, which means accompanied by entry restrictions and by measures that prevent increase in effort (seepage) by investments in unrestricted production factors. Further, if the fishermen’s behaviour is short sighted (myopic), and if the fishermen are taken by surprise by the allocation of the subsidy, important criteria for successful implementation of decommissioning grants are present: [26–28]. The effect is shown in Figure 5.3. Because of the increased opportunity costs of the fishermen, the supply curve after decommissioning moves upwards and the backward bending shape diminishes. The result is increased supply and lower prices.

Decommissioning grants aim at reducing total fishing effort. Fishermen are, in a sense, given better opportunities outside of fishing with such grants. Thus, the decommissioning subsidy will, in this way, increase the opportunity cost of fishing, which is reflected by the supply after decommissioning shown in Fig. 5.3. PricewaterhouseCoopers [24, chapter 8] argue that this increase in opportunity costs may be regarded as a tax on effort.

In a single country, single species, single fleet case, supply will increase, consumer prices will decrease, fishing effort will decrease and fish stock abundance will increase. In a two-country case (exporting and importing) where the exporting countries apply decommissioning grants, the result could be summarised as in Table 5.2.

It could be argued that seepage is extremely difficult to avoid, and that it will eventually lead to an increase in effort. Without entry restrictions and strict enforcement this will certainly happen, and even in cases where entry restrictions are enforced some seepage may take place. It is, however, hard to envisage that such developments will outweigh the economic benefits of the effort reduction. On the other hand, if the costs of reducing effort and enforcing effort reductions are higher than the gains, the method will lead to welfare loss for society.

Several authors have evaluated decommissioning programmes implemented in different countries. Woodrow [29] described the Canadian buyback and social adjustment

### Table 5.2 Impact of decommissioning scheme in export and import countries given that the stock is overexploited.

<table>
<thead>
<tr>
<th></th>
<th>Exporting economy</th>
<th>Importing economy</th>
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</thead>
<tbody>
<tr>
<td>Consumers</td>
<td>Price decreases and quantity increases</td>
<td>Price decreases and consumption increases.</td>
</tr>
<tr>
<td>Government Expenditure</td>
<td>Depends on cost of implementation and efficiency of decommissioning scheme</td>
<td>No direct effect</td>
</tr>
<tr>
<td>Industry</td>
<td>Effort decreases. Employment decreases. Price decreases. Harvest may increase.</td>
<td>Effort decreases. Harvest may increase.</td>
</tr>
<tr>
<td>Natural Resource</td>
<td>Effort decreases, stock increases</td>
<td>Effort decreases, stock increases.</td>
</tr>
</tbody>
</table>

Source: PricewaterhouseCoopers [24], Exhibit 8–2.
Financial instruments

programmes implemented during the northern cod moratorium. Holland et al. [26] examined several experiences in Canada, the USA, Australia, Norway, and in the European Union, including Denmark, the Netherlands, and the United Kingdom.

The mechanics of different programmes varies widely. Some programmes, which aimed at reducing fishing power quickly, have established eligibility requirements, methods of compensation, and directed funding toward the more productive fishermen. Other programmes, which placed more concern in maximising long-run capacity reduction, have targeted under-utilised licenses, latent effort and marginal fishermen to avoid the scenario that the successful applicants could purchase another fishing license, re-enter the fishery, and still use the same vessel.

Another distinction between programmes is found in the conditions for reuse of the vessel, gear, and human capital associated with a retired license. These conditions were put in place in order to prevent their use in upgrading the remaining fleet and their spill over into other fisheries, which may already be overcapitalised. However, most of the considered programmes put no or weak conditions on the reuse of licenses and vessels. The improvement of long-term resource stability and profitability through decommissioning programmes requires not only restricting new entry of vessels, but also avoiding increases in effective fishing effort derived from various forms of input stuffing by remaining fishermen. Thus, constraining the growth of effective effort requires that very strict effort restrictions are introduced and maintained.

Most subsidies do not operate in isolation: they are part of a package of economic support and policy measures that influence the performance of the fishing sector. As such, subsidies are not necessarily the only or most important factor determining sector performance [12]. Non-subsidy factors, including the effectiveness of the fisheries management system, changing market forces, changes in the environment, the effectiveness of sector governance, are sometimes at least as important. Most analysts agree that a well functioning fisheries management system in terms of effective sector governance will minimise the need for and effects of most fisheries subsidies. It is, therefore, useful to look at the role of the government in the sector in some detail, define when legitimate government support ends and should be called a subsidy, and assess what may be the optimal level of public support in legitimate public expenditures and subsidies to the sector as the sector develops and matures.

5.1.4 Government positions on subsidies

The World Wildlife Fund (WWF) [30–38] has, for some years, paid strong attention to the use of financial transfers in the fishing industry. However, WWF is a non-governmental organisation (NGO). Indeed, in terms of governments, although initiatives were taken to put fisheries subsidies on the agenda of the WTO Ministerial Conference in Seattle, USA, in 1999, it could be argued that fisheries subsidies were not treated separately and extensively until the WTO Ministerial Conference in Doha, Qatar, in 2002. Iceland recalled that paragraph 28 of the Doha Ministerial Declaration committed WTO Members to improve discipline on fisheries subsidies. These negotiations were taking place in the WTO Negotiating Group on Rules, but the US, New Zealand, Chile, Iceland and Australia argued that the WTO Committee of Trade and Environment should continue
to play a role in gathering and analysing information. Argentina, Jamaica, Peru, the Philippines and others noted the need for further analysis to take into account the special conditions in developing countries. Japan emphasised the need for a holistic approach to sustainable fisheries management, taking into account the demand driven causes of fish stock depletion. Korea and the EU maintained that the duplication of discussions on fisheries in WTO bodies should be avoided [39].

With reference to a paper by Munro and Sumaila [40] Korea noted that real world fisheries management was more complex than suggested by the theoretical analysis in this research paper. The main cause of overfishing was poor management of fish stocks and open access fisheries management systems, which create a powerful economic incentive for over-fishing. Korea argued that overcapacity will occur regardless of subsidies, while Japan argued that further studies on the effects of fisheries subsidies in other international organisations, including the FAO, were vital to address this issue in a scientific and factual manner. Thus, they argued that the discussion of financial instruments in fisheries should take place outside the WTO.

Overall, then, while a long list of countries were in favour of subjecting fisheries subsidies to negotiations within the WTO, Korea, Japan and the EU were more or less against it. Since then, the reform of the EU Common Fisheries Policy (CFP), which came into force in 2003, meant that provisions have been made for reducing Europe’s (harmful) fisheries subsidies, leaving Korea and Japan as the prime advocates for the holistic approach to fisheries subsidies, which is to include management aspects as well.

Following the Doha Conference a number of position papers have been forwarded. The positions of the above mentioned countries have not changed but a number of African, Caribbean and Pacific (ACP) countries have pointed out the need for specific crafting of measures with respect to developing countries that are highly dependent on fisheries.

5.2 THE KNOWLEDGE BASE

The fisheries knowledge base has two general kinds of sources: the research-based knowledge that is produced by the institutions of formal science; and the experience-based local (ecological and social) knowledge produced by fishery workers as they go about their activities. When the knowledge needed for fisheries management is contributed to and shared by several stakeholders, who can supplement the knowledge produced by scientists, there is a need for their participation in decision-making. This chapter will focus mainly on the research-based knowledge framework, although a broad knowledge of the local context, based on more detailed information and more continuous observation than is usually available from research-based sources, is very useful and important.

Although research-based knowledge has typically been a responsibility of the government, in the last decade the quota owners’ organisations of some countries with market-based fisheries management systems, such as New Zealand, have been collaborating and even driving and financing fisheries research-based knowledge, which, in some cases, is being purchased by the Ministry of Fisheries.

Historically, scientists have been providing advice on the basis of a ‘single stock or a single species’ in the form of fishing mortality limits and associated TAC options, but
the decision-making needs of the industry and managers are pushing scientists to use the ‘fishery’ as the unit of reference for advice. The term ‘fishery’ must be understood as a complex network, which includes economic, social, biological and technical aspects that are linked together.

The ‘fishery approach’ implies an evolution from using rather simple models towards increasingly complex ones—for example, the mixed-species TAC approach, the precautionary approach and the ecosystem approach—that tend to become broader, more multi-disciplined and include more uncertainties and mutual influences.

In particular, the fishery approach to research-based knowledge is holistic and requires that advice be fitted to a social unit rather than a biological one: see Wilson and Delaney [41]. Its central bio-economic framework is more appropriate than the traditional single stock approach for managerial purposes, but demands substantial data for modelling purposes.

Literature on pollution control as well as on fisheries economics has shown that Pigouvian taxes and transferable permits or quotas (ITQ) are equivalent in terms of economic efficiency under full information, since both instruments secure a first-best optimum. Nevertheless, the choice between price and quantity regulations under imperfect information depends on the slopes of the marginal revenue and marginal cost functions, see Weitzman [42], such that, if the marginal revenue function is ‘flat’ and the marginal cost function is ‘steep’, price regulation is preferred over quantity regulation—and vice versa. However, Jensen and Vestergaard [43] showed that Weitzman’s result holds for schooling fisheries (pelagic fisheries) but not for a fishery where the fish stock abundance influences the cost of fishing. Weitzman [44] also incorporated “ecological uncertainty” into the analysis as imperfect information on the fish stock-recruitment relation, and showed that taxes were preferred over ITQs in this case.

Taxes are hard to compute, and in many societies taxes are hard to change each year in response to changes in resource stock conditions. However, even if these arguments are accepted as correct, the analysis shows that taxes are preferred over ITQs under a scenario of ecological uncertainty and, when there is economic uncertainty, for some fisheries without stock effects on the cost side, the total benefit will be higher with taxes. Furthermore, price and quantity regulations have different distributional effects, because society normally collects the resource rent through the taxes, while the fishermen normally collect the resource rent in an ITQ system.

Arnason [7] developed a general ecosystem fisheries model under full information and examined economic methods for managing ecosystem fisheries, such as: corrective taxes or subsidies on harvest, equivalent to the shadow value of biomasses, and an appropriately designed ITQ system. The imposition of optimal unit harvesting taxes or subsidies as a management method presents serious practical problems, because they are demanding in terms of information and computation. First, one tax rate for every fish stock in the ecosystem is required, which implies a high number of taxes. Second, optimal taxes are dynamic, which means they must be continuously adjusted over time. Third, to obtain the pertinent instantaneous information about the growth of the fish stocks, the harvesting functions, and the profit functions, which are needed for the tax calculations, insurmountable problems arise for the fisheries manager.
Pradhan and Chaudhuri [8] presented a dynamic reaction model reflecting the dynamic interaction between the perceived resource rent and the fishing effort in a fishery that deals with a problem of non-selective harvesting of two competing species.

Jensen and Vestergaard [10] focused on the problems of illegal landings and discards in a system with imperfect information about the size of the individual catches caused by the problems with monitoring landings and observing fishermen’s behaviour at sea. If the costs of fishing and the stock size, for example, are observable from fish stock assessments, a fish stock tax/subsidy mechanism that uses the stock size as the tax base is proposed as a solution to the problem. The analysis, which is based on a single-species approach, where the total quota is distributed to fishermen as individual quotas, concludes that when the actual fish stock is above the optimal stock, fishermen receive a subsidy equal to the difference in stocks multiplied by a variable individual subsidy rate, and when the actual fish stock is below the optimal stock, society places a tax on the fishermen.

Herrera [45] compares three bycatch control instruments—taxes/subsidies on bycatch, trip limits, and value based trip quotas—with a scenario of no restrictions on by-catch in a two-fish stock system that combines stochastic by-catch rates with discrepancies between ex-vessel prices and social (shadow) prices in a dynamic model. The main conclusion of this study is the dominance, from an economic efficiency point of view, of the tax instruments over the other three scenarios to eliminate discarding.

Garza-Gil et al. [5] developed a bio-economic optimisation model for the European southern hake stock to illustrate the way that a tax on effort, applied to a multi-fleet fishery composed of trawlers, vessels using longlines (hooks), and vessels using gillnets, could lead to a socially optimal exploitation of the hake resource. They demonstrate that different technologies operating simultaneously in the fishery lead to different equilibrium levels of tax on effort for each fleet, based on the differences in the marginal productivity of each fleet. In consequence, the tax rate is greater on the trawler fleet than on the other fleets because the effort from trawlers has a stronger negative effect on the hake biomass. The analysis was not based taxes on landings because these are different from catches in fisheries where the quantity discarded is not known.

To summarise the tax instruments described above: (a) a tax or subsidy on catches; (b) a tax on the landed fish; (c) a tax based on the stock size; (d) a tax on the difference between the ex-vessel price and the social price; and (e) a tax on effort, it is observed that the last four alternatives have been used in cases where society has imperfect information about individual catches due to problems of illegal landings and discards.

When illegal landings and discards are not present, the implementation of either a tax on the value of landings (gross revenues) or on profits of each fleet segment does not cause any type of distorting incentives to the fishing firms. The use of an effort related tax, for example a tax on operating costs, can potentially have distorting impacts and cause efficiency losses, because vessel operators would attempt to substitute inputs subject to taxation with inputs that are not subject to taxation. The same behaviour will take place with regard to catches, where species not subject to taxes will substitute species subject to taxes. Finally, license fees are not considered ideal fisheries management instruments because this type of tax has no relationship with the actual catches of the vessel, and because it is fixed on an annual basis. When substantial license fees are charged over and above the administrative costs of vessel registration, they have an impact on profits and,
thus, on fishing capacity by making investments in fishing seem less attractive. In the short
and medium term, the industry can react by fishing more intensively (thereby increasing
fishing effort) to make up for the higher costs or overexploiting some high value stocks
if there are no restrictions on the kind of fisheries the vessel can participate in. In this
scenario, it would only be in the longer term (when some firms become economically
unfeasible and have to leave the fishery) that the fee would have the intended impact of
reducing fishing capacity and effort. Matthíasson [6] discusses arguments for introducing
a fishing fee to Icelandic fleets as a way of extracting the resource rent. Most of his
arguments are favourable to their implementation if they are kept neutral—that is, if they
do not affect the use of factors of production for other sectors of industry.

The two types of bio-economic methodologies or decision tools, used more and more
often in the fisheries literature for managerial or planning purposes, are the optimisation
and the numerical simulation models in either deterministic and stochastic approaches.
Brekke and Moxnes [46] found in experiments that both tools had approximately the
same positive effect on management, but the models were useful for different reasons.
The optimisation models helped to identify appropriate target stocks and policies more
efficiently than managing without this tool, and the simulation models helped to avoid
destabilising overreactions. Thus, in their opinion, these two tools are complementary,
rather than alternatives. For a European review of operational bio-economic models, see
Frost and Kjaersgaard [47].

Risk analysis has also increasingly been used in practical settings to assess the implica-
tions of alternative scenarios in the presence of uncertainty arising from randomness and
parameter uncertainty. Nevertheless structural uncertainties, which reflect a basic lack of
knowledge about the nature of the fishery system, its components, dynamics and internal
interactions, are more difficult to deal with in a quantitative analysis framework. As
knowledge is often limited, the management framework must provide acceptable results
under these circumstances.

It is possible to assess the impact of the subsidies on prices, (intervention prices), and
on operating costs, such as subsidies on fuel, often demanded by the industry in times
of economic hardship on the whole system (industry’s earnings, government’s treasury
income, trade, and stocks). For example, the decommissioning subsidy is an incentive
or compensation for the withdrawal of fishing vessels from the fishery. The level of
compensation acceptable to a fishing firm will depend on the expected net earnings of
the vessel during its remaining lifetime and the value of the entitlement to exploit the
fishery in the future.

The sources of both kinds of financial transfers are numerous and range from supra-
governmental bodies to regional and local authorities; while their targets range from
direct industry transfers to indirect transfers in terms of public support for information
gathering, research, and quality control. This diversity means that methods of analysis
vary. In each case the methodology of analysis that can be used is entirely dependent on
the data available and its level of aggregation. As such it is important to underline several
common problems that relate to the economic data.

First, the industry’s costs and revenues, needed in order to analyse the influence of
the financial transfer programmes on the profitability of each segment of the fishing
industry in the short and medium-term, are usually neither available nor easy to obtain.
Thus, empirical analysis is often based on aggregated costs and income information from officially published statements.

Second, the public sector’s costs and revenues gathered in the public accounts are very often dispersed among governmental departments and different public bodies responsible for the programmes. Sometimes, the available information is simply inadequate or not designed to permit proper empirical analysis of the various financial transfer programmes, whose impact becomes impossible to quantify in any exact way. Moreover, in a number of countries with decentralised systems the transfer programmes are shared among two or more levels of government, from local to national, and the potentially significant expenditures by sub-national authorities may be, in some cases, not easy to obtain. Further, even when information is provided, it tends to vary considerably in terms of both usability and detail.

Third, public and industry data at the international, national and sub-national levels is necessary in order to make comparative assessment analysis, which requires some uniform criteria to assist in studying fisheries subsidies. Fortunately, there have been some advances in this area (works of OECD, FAO, WTO).

5.3 CHARGES AND SUBSIDIES AROUND THE WORLD

The purpose of this section is to throw light on the diversity and complexity of financial instruments, used mainly by the developed world with the aim of pursuing the sustainable exploitation of fisheries resources. In this context, it also demonstrates the prominent extension of subsidies (see Appendix 5.A.1), as well as the differences found between the theoretical and the empirical settings.

It is outside the scope of this discussion to analyse and compare the effectiveness of the different measures and systems examined around the world, as well as to explore to what extent each country makes use of harmful instruments.

5.3.1 EU instruments

The Treaty establishing the European Community entails that the Community exercises its exclusive competence in the areas of conservation, management and exploitation of living aquatic resources. The objectives of the Common Fisheries Policy (CFP) have been reviewed in this decade and now focus more on the sustainable exploitation of living aquatic resources. The management instruments of the reformed CFP, which came into effect on the 1st of January 2003 [48] are: (a) TACs and quotas decided annually by the Council on catch and fishing effort limits and on the allocation of fishing opportunities among Member States; (b) long-term plans setting objectives for specific stocks that need recovery; (c) a fleet management policy to overcome the challenge posed by the chronic overcapacity of the EU fleet; and (d) the common organisation of the market for fishery and aquaculture products to secure price stability and an optimal balance between supply and demand.

The proposals on fisheries management made by the European Commission are the result of a decision-making process based on scientific advice, which integrates biological,
economic, environmental, social and technical variables, and on a broad involvement of stakeholders.

The Advisory Committee on Fishery Management (ACFM) of International Council for the Exploration of the Sea (ICES) is the most important official scientific body providing advice to the Commission of the EU through DG-Fish. The Commission also has its own scientific advisory Committee—the Scientific, Technical and Economic Committee for Fisheries (STECF)—the role of which is to address management issues and, in particular, to examine the stock assessments carried out by national research institutes on behalf of ICES. STECF bases its recommendations on reports carried out by two subgroups: the SGRST (subgroup on Reviews on Stocks) and the SGECA (subgroup on Economic Assessment). The SGECA incorporates the annual TAC proposals and long-term projections (for example single species, mixed fishery, and management plans scenarios) into an economic model to predict the economic consequences for a selected number of fleet segments in each Member State.

The Commission consults regularly with the representatives of various branches of the industry and with other stakeholders within the framework of the Advisory Committee on Fisheries and Aquaculture (ACFA). However, despite the existence of this large-scale participation, the industry often complains about being excluded from the TAC-setting process. The recently established Regional Advisory Councils (RACs), composed of fishermen and other representatives of other interest groups operating in the sea, also advise the European Commission on matters relating to the fisheries management of certain species or fishing zones.

The European Member States are responsible for the implementation of the CFP in their waters and territory and also for their vessels operating outside the Community waters. Thus, once the fishing quotas have been established for each country, each Member State decides on the method of allocating the fishing opportunities assigned to that Member State in accordance with Community law. In relation to the new fleet policy, the Member States have responsibility for matching fishing capacity to fishing opportunities and must manage the entries and exits of the fleet [21, 49]; Hatcher, 2000). The European Commission evaluates and controls the application of the CFP by the Member States.

Most EU countries use input controls (for example licenses, individual effort quotas, and regulations on gear and vessel specifications) and technical measures (time/area closures, etc.) to manage their fisheries. However, in keeping with the European TAC approach, many states use economic instruments that control the output from fishing, such as individual quotas and individual transferable quotas.

Under the terms of the Treaty of the European Union, the CFP must be consistent with other EU policies and, importantly, economic growth and cohesion are among its general goals. The use of subsidies in the Union reflects the situation that high productivity rates and income increases in certain regions often run parallel with lower productivity rates and increases in income in other regions. Subsidies to the fishing industry, usually located in remote and/or less developed areas, have been a measure to secure cohesion. It is now realised that subsidies could be harmful with respect to overexploitation of fish stocks, and provisions have been made to put renewed focus on the conflicting objectives as regards cohesion, on the one hand, and protection of the environment including the fish stocks, on the other [50–53].
The subsidies allocated to the fishing industry are part of the four EU Structural Funds [32]:

- Financial Instrument for Fisheries Guidance (FIFG),
- Regional Development Fund (ERDF),
- Social Fund (ESF) and
- Agriculture Guidance and Guarantee Fund (EAGGF).

In 1999, new regulations for the revision of these Community Structural Funds were approved. The objective of this reform was to reduce the development gaps between regions and to increase the effectiveness of the Funds. After the reform of the CFP the general priorities of support of the EU Structural Funds have been reduced to three:

Objective 1 covers regions whose development is considered to be lagging behind (with a gross domestic product per head of population of less than 75% of the Community average), or that are located in remote areas: for example, French overseas departments, the Azores, Madeira and the Canary Islands. Coastal areas and fisheries activities located in these regions can obtain as much as 75% of aid under the FIFG. Furthermore they can access the European Regional Development Fund (ERDF) and the European Social Fund (ESF).

Objective 2 covers areas undergoing economic and social conversion, including depressed areas dependent on fishing. The areas are selected by Member States according to two criteria: (a) the share of jobs in the sector compared to total employment; and (b) the amount of job losses in the sector as a result of restructuring.

Objective 3 focuses on supporting the adaptation and modernisation of policies and systems of education, training and employment and operates in areas not covered by Objective 1.

The FIFG is by far the main instrument with respect to the fishing sector, since it provides financial support to achieving the main objectives of the Common Fisheries Policy and provides incentives to develop measures accompanying the CFP in the framework of the cohesion policy. Taking into account the results of its previous funding programme for 1994–1999, there have been a number of changes, in particular with respect to subsidies that are environmentally harmful. The areas of assistance, measures, and changes can be summarised in the following eight items [50, 51].

Permanent cessation of fishing activities, including scrapping, export to a third country or assignment to activities other than fishing. These have been retained subject to conditions regarding the age and tonnage of the vessels concerned.

Creation and operation of joint ventures have been reviewed and tightened in order to ensure an effective reduction in the fishing effort on the resources previously exploited by the vessel concerned and to guarantee that international law is not infringed.

Renewal and modernisation of the fishing fleet: The general principle is that public funding should not contribute to an increase fishing capacity. To be allowed to allocate
public support, Member States must establish permanent mechanisms for monitoring fleet renewal and modernisation and prove that the development of their fleet does not exceed the annual objectives fixed in the Multi Annual Guidance Programme (MAGP) for the overall fleet and for the segment concerned.

Support to investment in Protection & development of aquatic resources, Aquaculture, Fishing port facilities and Processing and marketing. Most of the previous measures have been retained, including aids to the producer sector, albeit within a stricter control framework.

Small scale coastal fishing: Considering the importance of fisheries carried out by vessels of an overall length of less than 12 m in a number of regions, and their contribution to employment, they benefit from specific measures, such as: (a) exceptions in the fleet renewal programme and; (b) a lump sum premium\(^4\) when a number of vessel owners or members of a fishing family put in place an integrated collective project to improve the structure of their fishing activities.

Socio-economic measures formerly available have been maintained and a new measure has been added to help young fishermen invest in their first vessel. The measures available for fishermen are: (a) co-financing of national early retirement schemes, subject to specific conditions relating to age\(^5\) and to the length of time spent as a fisherman\(^6\); (b) allocation of individual compensatory payments\(^7\) in case of redundancy due to the permanent cessation of fishing activities of the vessel; (c) allocation of non-renewable individual compensatory payments\(^8\) to help fishermen, who have been employed for at least five years in the fishing industry, retrain or diversify their activities outside sea fisheries. These payments may be combined with redundancy payments; and (d) granting of individual premiums to fishermen under 35 years who have worked as fishermen for at least five years or who have followed vocational training and who, for the first time, become owner of a fishing vessel\(^9\). The vessel must be between 10 and 20 years old and of a length ranging from 7 to 24 m. The maximum grant will be 10% of the purchase cost or €50,000.

Measures taken by the industry encouraging the industry to undertake activities aimed at strengthening the fishing sector have been retained and have been given more emphasis: for example, collective schemes for the promotion of and search for new markets and uses for the products. This includes quality certification, product labelling, promotion campaigns, market surveys, fairs and exhibitions, marketing advice and support. Collective schemes such as Producers’ Organisations, Fishermen’s Organisations, Associations or Groups aiming to improve the quality of their products are also included, as well as innovative measures and technical assistance: for example, with exploratory fishing, training

\(^4\) Maximum €150,000 per integrated collective project.
\(^5\) Beneficiaries must be within 10 years of the legal retirement age or aged at least 55.
\(^6\) At least ten years.
\(^7\) Maximum €10,000.
\(^8\) Maximum €50,000.
\(^9\) The transfer of ownership must not take place between members of the same family up to the second degree.
programmes, exchange of know-how and measures promoting equality of employment opportunities between men and women working in the sector.

*Temporary cessation of activity and other financial compensations* to fishermen and owners of vessels forced to cease their activities temporarily due to: (a) unforeseeable circumstances; (b) non-renewal or suspension of a Fisheries Agreement between the Community and a Third Country or within the framework of a Regional Fisheries Organisation; (c) introduction of a scheme for the recovery of a species threatened with depletion and; (d) technical restrictions on the use of specific fishing gear or techniques decided by the Council of Fisheries Ministers.

The maximum rate of assistance from the EU (FIFG) for investment in private businesses is brought down to 35% for regions under objective 1 and to 15% elsewhere\(^{10}\). The minimum national contribution has been maintained at 5%. The private sector must carry the remainder of the investment costs. Support for the renewal and modernisation of the fleet is an exception to this rule, since it requires that the minimum participation by the beneficiary be at the same level regardless of the region concerned (80%).

For collective infrastructures and premiums the maximum EU assistance rates of 75% (objective 1) and 50% (elsewhere) have been maintained as well as the minimum rate of National assistance (25% or 50%). Subsidies provided from the structural funds, including FIFG, are also subject to the subsidiarity principle, implying that it is the responsibility of the Member State to design, implement and co-finance the subsidy programme. The procedure is laid out in fisheries development plans for each Member State. There are two types of development plans: (a) The Community Support Framework (CSF) and the Operational Programmes (OP) related to a country, or a group of regions within a country, funding from which is eligible under Objective 1, where the CSF describes the social and economic context of the country or regions covered by the Structural Funds, sets out development priorities and targets to be attained, and provides for financial management, monitoring, evaluation and control systems, and the OP lists the various priorities of a CSF for a particular region or a particular development sector; and (b) The Single Programming Documents (SPDs), which are applied elsewhere and feature aspects of both a CSF and an OP.

The preparation and administration of these development plans are subject to cooperation between the Member State and the Commission. The broad priorities of a each programme are identified in cooperation with the Commission, but the choice of measures and practical projects is the sole responsibility of the Member States. The Commission holds the authority to approve the plans.

The commitment and necessary consensus between the various parties involved (Commission, State and other authorities or partner bodies designated by the national authorities, such as relevant public authorities at regional and local level, or economic and social agents, or other appropriate bodies) is achieved through ongoing dialogue throughout the programming process. These parties are also invited to monitor the implementation and

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\(^{10}\) As compared to 50% and 30% in the previous programme.
financing of the programmes by taking part in monitoring committees and in the various evaluation exercises.

Member States have responsibility for managing expenditure. In particular, they need to demonstrate that suitable structures are in place in each region or Member State to administer the funds, but also to monitor expenditure, and evaluate how effective and efficient programmes are. The managing authorities (national or regional authority or an intermediary body entrusted with a public interest mission) are also selected by the Member States. They are responsible for organising the collection of financial, environmental and statistical data on the programme being managed.

The beneficiaries of assistance receive funds from a payment authority designated by the Member State who play a central role by reimbursing expenditure carried out and by checking the eligibility of this expenditure, before payment is made. To the extent that this authority must certify all expenditure included in requests for payment, it assumes a large part of the State’s responsibility to ensure that the payments are legal and correct. Furthermore, it must keep accounts of expenditure and verifiable evidence. In practice, the managing and payment authorities can be part of the same governmental body, but the principle of separating functions implies that they must belong to distinct departments.

The Member States also set up a monitoring committee for each programme (OP and SPD) to ensure the quality and effectiveness of the implementation of assistance. The monitoring committee is in close contact with the European Commission, which participates in its discussions on a consultative basis.

Evaluation exercises take place in the mid-term and at the end of the programming period. The mid-term evaluation is organised by the managing authority but is carried out by an independent assessor. It is designed to examine the initial results of the assistance, the use of financial resources and the operation of monitoring and implementation. Once the assistance comes to an end, an ex post evaluation is carried out under the main responsibility of the Commission. This ex post evaluation, also conducted by an independent assessor, highlights the factors contributing to success or failure in implementation in order to optimise the assistance provided in the future. The managing authority is responsible for establishing a reliable system for collecting statistical, financial and environmental data on the programme being managed as well as for evaluation purposes.

An ambitious programme has been agreed in the European Union regarding compulsory biological and economic data collection at the Member State level and submission of the information to the Community level [54, 55]. This programme aims to harmonise and expand the existing data collection that already takes place in many Member States. Further actions have been taken to establish a scientific framework that would make it possible to combine the theoretical experience within biology and economics with the comprehensive data collection programme. In this respect, the EU seems to be at the cutting edge on a worldwide scale. However, there remains room for improvements relating to a more holistic use of this information with respect to trade distortions, environmentally damaging consequences, and distributional effects (cohesion).

The total budget or maximum public spending (FIFG + Member State), in both the harvesting and the processing industry, is €5661 million for the seven years (2000–2006) programmed period.
However, most European countries, with the exception of Greece and Netherlands, have also granted their own national and regional aids to complement (or to reinforce) the measures derived from the general framework of EU structural aids. This, broadly defined, ‘State Aid’ [32], can include all forms of aid made available to the sector, such as grants, interest subsidies, tax credits and other tax measures, and reductions in social security contributions. Cofrepêche [56] measured the total amount of State Aid given by the Member States of the EU to their fishing industries during the period 1994–98 at €377 million. This amount represents 12% of total aids given in the FIFG program 1994–1999. Italy was the Member State that used the most of this type of aids (58%), followed by France (11%), Germany (10%), Spain (8%) and UK (7%).

The CFP does not apply any charge scheme to the fishing industry. Nevertheless, some EU Member States have introduced charge measures (for example access/entry taxes, input taxes, output taxes, and fines) in their fisheries sectors, but there is very little information available about these. Charges to recover the management costs of the fishing industry are not being considered by either the EU or the Member States.

5.3.2 Financial instruments in operation outside the EU

5.3.2.1 Different management systems

New Zealand, Australia and Iceland have completed restructuring their fishing activities, using ITQs to manage most of their fisheries, and have reduced net levels of transfer to the sector to a minimum [57, 58]. The amounts of public financial transfers are small and well defined. The highest exponent of this is New Zealand where expenditure is limited to General Services for the harvesting sector and where 43% to 67% of this expenditure is recovered from its commercial fisheries. This expenditure amounts to close to 100% of the costs attributed to management.

Norway regulates its coastal fishing vessels through annual permits, while the rest of the commercial fishing fleet is subject to a licensing system. Closed access on a stock-by-stock basis has been implemented to such an extent that 90% of the catch value comes from access regulated fisheries. Input regulations have also been applied to limit the fishing effort. Alongside these are structure/rights regulations aimed at reducing the fishing capacity of vessel groups that coexist. A Unit Quota System has been established for the offshore fishing fleet and the coastal fishing fleet. A Quota Exchange System (QES) for vessels of less than 28 m in length allowing two vessel owners to team up and fish both quotas on one vessel for three out of five years was tested in 2004. Depending on the result of this trial the QES may be extended to the rest of the fleet. The aim of this system is to improve vessel profitability and to incentivise a reduction in fleet capacity. In most fisheries a TAC is set, which constitutes a national quota that is divided between vessel groups in a manner that results in a certain competition between the vessels in the group. In addition, period quotas, trip quotas, and quotas of days at sea constitute input controls and are used to limit the output in some fisheries. These quotas are not, in principle, transferable but the fact that quotas are linked to specific vessels means that the price of a vessel traded includes the value of its quota or, more specifically, the right to fish.
Canada, the United States and other countries presently have relatively high subsidy levels and vessel replacement rules to control growth in capacity. They have introduced ITQs in some of their fisheries. The fisheries management regimes of Canada and the USA are quite complex because different schemes coexist (quota and non-quota fisheries). They apply various input controls (limited licenses, individual effort quotas, gear and vessel restrictions), output controls (TACs, escapement/recruitment targets, individual quotas and catches per trip or per short period of time), and technical measures (size/sex selectivity, time/area closures). The instruments used in Canada take the form of enterprise allocations and individual quotas. Transfers of quotas, in the form of ITQs or individual transferable vessel quotas take place, if agreed by the government and industry participants in the fishery. At a more aggregated level, individual community groups manage community based quotas and temporary quota transfers are permitted between communities. These rights-based systems have gained increasing acceptance in Canada and, in 2000, 19% of commercial licenses/permits issued for all major species were under a variety of rights-based systems.

Japan has three types of fisheries: coastal fisheries, offshore fisheries and distant water fisheries; and employs multiple layers of resource management procedures. A vessel registration system closely monitors the number and the gross tonnage of the fishing vessels. A national fishery license is required to operate on a nationwide scale or in international waters. The number of licenses are strictly limited and closely controlled by the government to control fishing capacity. A prefectural license is required for fisheries operating on a regional scale. The central government determines the upper ceiling of the number of fishing vessels licensed by Governors in order to set the maximum overall catch limit. A catch control through TACs is imposed for seven major fishery species. An upper limit on the number of fishing days and the number of operating vessels in a specific area within the EEZ was established through a total allowable effort system. Moreover, in the coastal areas a traditional rights-based management system is operating, which overlaps the management measures listed above. In the coastal fisheries a group of fishermen (fishery cooperative associations) traditionally assume exclusive rights for operating certain fisheries and, thus, assume all the responsibility for long-term sustainability of resources. Rights to fish are not transferable. There are three types of fishing rights. First, the joint fishery right is based on common ownership systems of local fishing grounds: it is granted only to fishery cooperatives, the members of which use the license on an individual basis. As a part of the implementation of fishing rights input control regulations on fishing seasons, areas and gear use are put in place by individual cooperatives. Second, the demarcated fishery right grants permission to engage in aquaculture, for which fishery cooperatives have the priority access. Third, the set net fishery right is in place for those fishermen targeting, for example, salmon, and yellow tail tuna, among other species; the self-supporting fishery cooperatives with a strong dependence on fisheries have priority of access to these rights.

Korea’s fishery sector is experiencing a significant restructuring. The deep-sea fisheries need a license issued by the Ministry of Maritime Affairs and Fisheries, while local government issues the license for offshore fisheries. A TAC policy was introduced in 1999 on four species and, by 2003, nine species were subjected to TACs. Fishermen oriented co-management systems have extended responsibility and rights to fishermen to
take over the management of fishing grounds, fishery resources and harvesting, all with a
sense of co-ownership. By 2003, the government had designated 122 fishing villages with
small-scale fisheries as co-management communities. They are devising policies for both
the conservation of fishery resources and for enhancement by creating artificial habitats
for marine aquatic living species, in transition from catch fisheries to nurturing fisheries.

An overview of financial instruments for important OECD non-EU countries is pro-
vided in Table 5.3.

5.3.2.2 Public support

The use of public aid to the fishing industry has been a common practice especially in
the developed world (see Appendix 5.A.1). The objectives of such aid have been diverse:
including balancing fishery resources and their exploitation; strengthening the compet-
itiveness of fisheries enterprises; improving the market supply from capture fisheries

Table 5.3 Economic and financial instruments for marine fisheries in some OECD countries
(excluding EU members) in the period 1999–2002.

<table>
<thead>
<tr>
<th>Instrument</th>
<th>Coverage</th>
</tr>
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<tbody>
<tr>
<td>Iceland</td>
<td>(A) Individual transferable quotas (B) No fees (C) Public Financial Transfers</td>
</tr>
<tr>
<td></td>
<td>(A) All fisheries (B) – (C) 3% to 4% of landed value</td>
</tr>
<tr>
<td>Australia</td>
<td>(A) Individual transferable quotas (B) Fees (C) Public Financial Transfers</td>
</tr>
<tr>
<td></td>
<td>(A) 5 fisheries (more than 22 species) (B) 29% of management costs (C) Around 8% of landed value</td>
</tr>
<tr>
<td>New Zealand</td>
<td>(A) Individual transferable quotas (B) Fees</td>
</tr>
<tr>
<td></td>
<td>(A) 92% of landed volume from EEZ: 45 species/290 stocks (B) To recover management costs (43% to 67% of expenditure in General Services) (C) 13–19 USD million.</td>
</tr>
<tr>
<td>Norway</td>
<td>(A) Individual quotas (B) No fees (C) Public Financial Transfers</td>
</tr>
<tr>
<td></td>
<td>(A) Used for the most important fish stocks (B) – (C) 8% to 14% of landed value</td>
</tr>
<tr>
<td>United States</td>
<td>(A) Individual transferable quotas (B) No fees (C) Public Financial Transfers</td>
</tr>
<tr>
<td></td>
<td>(A) 3 fisheries: halibut/sablefish; wreckfish; surf clam/ocean quahog (B) – (C) 28% to 37% of landed value</td>
</tr>
<tr>
<td>Canada</td>
<td>(A) Individual transferable quotas (B) No fees (C) Public Financial Transfers</td>
</tr>
<tr>
<td></td>
<td>(A) 50% landed value (B) – (C) 34% to 39% of landed value</td>
</tr>
</tbody>
</table>

(Source: OECD [59]).
and aquaculture products; and keeping small fishing communities alive. A few recent examples of subsidy schemes in selected countries are briefly presented in the following.

Canada uses license buyback, early retirement programmes, short-term income support, retraining and economic diversification to assist affected fisheries workers and communities.

The USA implements several subsidy schemes: buyback programs, capital construction funds and other tax programs, the fisheries obligation guarantee program, the Wallop-Breaux program for recreational fisheries and other programs, such as the ones related to disaster relief, small business administration, economic development administration, farm credit system, fisheries development, marketing & promotion programs. The Saltonstall-Kennedy Act; the USDA Food Aid Program; the USDA School Lunch Program; the Foreign Agriculture Service; the National Fish and Seafood Promotion Council and the Sea Grant College Program also provide public funds to the fisheries sector.

In Norway the problem of overcapacity is still present in the fishing fleet, and, therefore, grants for constructing new vessels are no longer provided. Decommissioning schemes are being used as an instrument to reduce the capacity of the fleet. Various schemes have been in effect more or less continuously for the last 40 years, including direct payments to compensate the low profitability of some fleet segments due to the joint effect of overcapacity and quotas. This support peaked in 1980 and has since been declining. In 2003, these payments amounted to NOK 70 million. In general terms, Norway has substantially reduced its public support to the sector during recent decades.

In Korea support is in place to improve infrastructures such as fishing ports and to promote fishing villages. Several subsidy programmes are related to fisheries development. These include artificial reefs, quality fry releasing, and marine ranches. Korea operates a decommissioning scheme to reduce over-capacity and implements social programmes.

In Japan funding is available for General Services (unspecified) for setting up new vessels, and in the form of direct payments for fishing fleet reduction and temporary cessations.

5.3.2.3 Charges

Charges are in place in a number of OECD countries, and a brief review of the measures of some of these countries, excluding EU countries, is presented.

Access or entry charges are found in countries where different management plans for fisheries coexist, such as fisheries under quota and non-quota systems. The licence fees charged to quota holders are different (e.g., calculated on a per tonne of quota basis in Canada, or a percentage of landed value in the USA) from the permit/licence fees charged in non-quota fisheries (e.g., a fixed amount based on a benchmark landed value in Canada, or a simple license fee in the USA): see OECD [17] and Hatcher and Pascoe [16].

Cost recovery charges are found in New Zealand and Australia, which charge their fishing industries with the objective of recovering the management costs directly related to fishing activities. Such cost recovery charges usually include administrative and monitoring costs and, partially, the costs attributed to research and data collection programmes. Governments pay for activities that may benefit the broader community, such as domestic fishing enforcement, foreign fishing compliance and non-commercial fisheries.
The information about cost recovery charges and the fees paid by the Canadian harvesting sector reveals an apparent contradiction. On the one hand, it is true that the income raised through these fees is not directly linked to the costs of management. However, on the other hand, it is also true that almost all rights-based fleets/fisheries are subject to industry paid observers at sea and dockside monitoring programmes, whereas other fleets/fisheries, subject to various degree of catch monitoring, do not pay for this service.

In the USA the income raised from the license/permit fees are not directly linked to the costs of management. However, this income is specifically earmarked for paying for management. Additional charges can be imposed in the USA (at the request of the industry) to fund buy-back schemes.

A new tax on the value of first-hand landings will partially fund a scheme for the decommissioning of vessels up to 15 m in Norway.

Use charges or penalties are used in almost all quota fisheries—in particular in rights-based systems—to penalise commercial fishermen who catch in excess of their annual catch entitlement (quota): see Frost and Jensen [60] for a Danish case. Although the rationale for this type of charge is the same everywhere, its implementation changes from place to place. In New Zealand, for instance, commercial fishers are liable for deemed values in terms of a fee set at a level established to encourage fishers to acquire annual catch entitlements to cover their catch or any catch in excess of their annual catch entitlement (ACE). Their payment can be satisfied by acquiring ACE or by paying the amount demanded.

Rent capture charges, established to provide a return to the community from the use of marine resources, are well-known within the theory. There are few references in the literature, however, to cases where this is applied. New Zealand managers imposed a resource rent charge when they started restructuring their fisheries using the quota management system (QMS) in 1986, but it was set at quite a low level to encourage fleet restructuring, and it only captured a small part of the resource rent. Norway has discussed proposals for a resource tax to cover the management costs paid by central government, but this has so far not been introduced.

5.4 CONCLUSION

The term ‘financial transfer’ covers both charges and subsidies. Financial transfers are strongly biased towards subsidies since their use is much more extensive. They cover actions between the government, including regional and local authorities, and the private sector. Financial transfers between private companies—for example vertical or horizontal integration—are disregarded. One definition that covers the subject reasonably well is:

Government action (or inaction) that modifies (by increasing or decreasing) the potential profits earned by the firm in the short, medium and long term.

The use of public aid to the fishing industry has been more intense and widespread in developed countries than it has been in the less developed world. Nevertheless, the extent
of such subsidies, in relation to the value of landed catches, has been declining in several
countries in recent years.

Charges or taxes, whether access/entry charges, such as annual license fees, or use
charges, such as effort taxes or taxes on landings, which impact the potential profit of
the firm, can be applied to help manage a fishery, to recover the costs of management,
or simply to raise public revenue by extracting some or all of the potential resource rent.

Management cost recovery and access charges are in place in a number of countries
around the world; but other types of charges directed at managing fisheries, such as an
effort tax or a landing tax are difficult to implement in practice and, thus, they are not
very commonly employed.

The above definition of financial transfer requires that the knowledge base be evaluated
in a broader bio-economic context. Therefore, the knowledge base must be evaluated with
respect to:

(1) Institutional structure
(2) Management procedures
(3) Models and data sources
(4) Scientific advice and communication of advice

5.4.1 Institutional structure

This chapter shows that management by use of financial instruments is extremely het-
erogeneous in the sense that it impacts upon fish resource exploitation in diverse ways
and often in an indirect manner. The sources of subsidies range from supra-governmental
bodies to regional and local authorities; while the targets of subsidies range from direct
industry transfers to indirect transfers in terms of public support for information gathering,
research, and quality control.

With reference to the definition given above, a distinction could be made between
transfer bodies and types of transfers such as taxes, levies, and fees. Transfers between
bodies require attention be paid to where they originate and where they go. This is not
always easy because there are many possible sources and recipients, and the magnitude
of the transfers may be difficult to estimate. On the other hand, transfers between bodies
are easier to track than other types of transfers because transfers between bodies rest on
observations and include no value judgements.

The other types of transfers are more difficult to handle because the various types of
transfers can be difficult to define, and the impact on private firms’ profit and behaviour
can lead either to distortion or restoration, in particular with respect to external effects
from society’s viewpoint. Addressing these other types of financial transfers requires that
management objectives and procedures be laid down, and that they are supplemented by
economic modelling to assess their impacts.

Subsidies to the fishing industry, specifically, have not, until recently attracted strong
attention. This is the case, first, with respect to trade distortions and, second, with respect
to environmental distortions, in terms of harmful overexploitation of fish stocks. There are
many international sources of information for these transfers, including the FAO, OECD,
APEC, World Bank, WWF, and, not least, the WTO. The WTO, together with OECD,
which are the formal bodies whose members are national governments, are regarded as providing the legal classifications of subsidies.

5.4.2 Management procedures

Although there is no final conclusion as to how management by use of financial transfers ought to be implemented, a way to address management procedures is to look at the classification of subsidies and rank the subsidies accordingly. The OECD fisheries-specific types of transfers [17] seem to be pertinent in this context:

(1) Fisheries infrastructure
(2) Management, research, enforcement and enhancement
(3) Access to other countries’ waters
(4) Decommissioning of vessels and license retirement
(5) Investment and modernisation
(6) Income support and employment insurance
(7) Taxation exemptions

These types of financial transfers cannot be judged harmful in advance with respect to either trade or environment. Their impact depends on the objective of the transfers rather than the type. It is theoretically possible to use both taxes and subsidies to pursue an efficient exploitation (by ensuring that the industry operates at an economically optimal level of effort, minimising negative environmental impacts), although the distributional consequences between groups of agents in the private sector, and between the private and the public sector, are different.

However, in practice, it would be extremely difficult to set either an effort tax or a tax on landings that alone would ensure an economically optimal fishery. In order to do so, it would be necessary to have full and detailed information about the cost structure of the industry, fish stocks, catch compositions including discards, and market prices, not just at one point in time but continuously. Moreover, in a multi-species fishery, setting correct (optimal) tax rates would be virtually impossible. However, less ambitious aims, leading to an improvement in the economic performance of the sector relative to the current one, could be sensible, and that could certainly be achieved by use of taxes.

5.4.3 Models and data sources

The bio-economic framework, which constitutes the basis for the analysis of the exploitation of fish stocks, requires three-dimensional variable information: (a) accurate biological data, knowledge about the population dynamics and status of the fishery exploitation, information about the impact of fishing on the fish stocks and, in a broader context, on ecosystems; (b) catch (including by-catch and discard) compositions and price information; and (c) fishing technologies and associated costs of each fleet segment. The financial transfers, understood as public incomes and costs, including the expenditures in General Services, such as monitoring, enforcement and research, and other policy instruments
used with management purposes, are added to the base model scenario and their impacts on the whole system are analysed.

Furthermore, economic information about the asset and liability structure of the fishing companies (balance sheets) should be regularly collected to analyse the impact of charge and subsidy schemes in operation in the industry. This information should also include the fishing effort and the usage of inputs per fleet segment and fishery, the industrial and fleet structure, as well as the structural changes in fishing-dependent communities and regions, taking into account the evolution of the processing and other connected industries.

Nevertheless, the strong demand for biological and economic information may be alleviated by use of proper models. Such models need to include biological and economic aspects and reflect that supply is backward-bending as a function of increasing fishing effort or fishing mortality. These characteristics of the fishing industry are different from almost all other industries and lead in many cases to conclusions that are different from the conclusion arrived at for other industries such as agriculture. However, only few (if any) models capable of addressing these issues exist.

The imposition of optimal unit harvesting taxes or subsidies as a management method presents serious practical problems, because they are demanding in terms of information and computation. Nevertheless, even in an environment of limited information appropriate taxes/subsidies can be designed. In consequence, taxes/subsidies remain an interesting option as a management tool because corrective taxes will almost always ensure an improved economic situation relative to open access, although the distributional consequences between groups of agents in the private sector and between the private and the public sector are different with both instruments.

Apart from the strictly bio-economic data, it is also necessary to have data regarding financial transfers, including the public costs of General Services (monitoring, enforcement, research, etc.). The sources of such data are, however, numerous, and the impact on the fishing industry’s competitive power and the environment depends on the objective of the transfers and their capacity to avoid or limit seepage. These data are best gathered from public sources (suppliers) rather than from the recipients. In particular national governments are good sources, and, on the international scale, OECD plays an important role. However, data collection in this area needs further development and reinforcement.

As the methodology of analysis that can be used is totally dependent on the data available and their aggregation level, it is important to underline the problems that frequently affect the economic data.

- The industry’s costs and revenues, needed in order to analyse the influence of the financial transfers programmes on the profitability of each segment of the fisheries industry in the short and medium term, are usually neither available nor easy to obtain. Thus, empirical analysis frequently uses aggregated costs and income data from officially published statements.
- The Public Sector’s costs and revenues gathered in the public accounts are very often dispersed among governmental departments and different public bodies responsible for the programmes. Sometimes the available information is simply inadequate or not designed to permit proper empirical analysis. Moreover, in a number of countries with
Astorkiza et al.

decentralised systems, the financial transfer programmes are shared among two or more levels of government and the expenditures by sub-national authorities may be significant and in some cases not easy to obtain.

- Public and industry’s data at the international, national, and sub-national levels is necessary in order to conduct comparative assessments. This requires uniform criteria to assist in the studying of fisheries subsidies. Fortunately there have been some advances in this area by the OECD, FAO, and WTO.

5.4.4 Scientific advice and communication of advice

A rather comprehensive theoretical literature is available regarding trade distortions and the correction of market failures. In the case of fisheries, the lack of useful economic information regarding financial transfers has made it difficult to implement the recommendations from the literature, not least because the economic repercussions in terms of distributional effects were unknown. An ambitious programme has been agreed in the European Union regarding compulsory biological and economic data collection at the Member State level and submission of the information to the Community level. This programme aims at harmonising and expanding the existing data collection that already takes place in many Member States. The use of financial transfers is well-documented by the EU’s structural policy, and, in this area, the EU seems to be at the forefront on a worldwide scale.

Room for improvements in this sphere is associated with a more holistic use of this information with respect to trade distortions, environmental disruption, and distributional effects, including cohesion. With respect to analyses, an institutional framework for scientific advice is not in place within the European Union. There is no experience in the world on a similar scale to the EU that could form the basis for further conclusions here. However, within the EU, the data collection programme, not least relating to economics, is not yet fully implemented; although, actions have been taken to establish a scientific framework that would make it possible to combine the theoretical experience within biology and economics with the comprehensive data collection programme.
Appendix 5.A.1

Table 5.A.1 Public financial transfers to the marine capture fisheries of OECD countries during 1999–2002 (USD million)

<table>
<thead>
<tr>
<th>MARINE CAPTURE FISHERIES</th>
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<th>CAN</th>
<th>ICEL</th>
<th>JAP</th>
<th>KOR</th>
<th>MEX</th>
<th>NZ</th>
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<td>121</td>
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<td>121</td>
<td>199</td>
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<td>49</td>
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<td>370</td>
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<td>NZ</td>
<td>NOR</td>
<td>POL</td>
<td>TURK</td>
<td>USA</td>
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<td>OECD(^2)</td>
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<tr>
<td>(\text{TT/Value Landings (%)})</td>
<td>3%</td>
<td>16%</td>
<td>11%</td>
<td>4%</td>
<td>37%</td>
<td>43%</td>
<td>45%</td>
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</tbody>
</table>

Blank means not available.


\(^2\) Excludes also: (A) Australia in 1999 and 2002. (B) Canada in 2002. (C) Mexico and Poland.

(Source: OECD [61, 62]).
REFERENCES


[38] WWF Germany Die Umwelt als zentrales Anligen der EU-Fischereipolitik. Position des WWF zur der Reform der Gemeinsamen Fischreipolitik der EU (2001).


6.1 INTRODUCTION

Traditionally, fisheries management has focused on protecting resources from overexploitation or on the recovery of already overexploited stocks. Total Allowable Catches (TACs) are a possible management tool to achieve this resource-protection goal, but have, in many cases, led to different ecological, social and economic (e.g., overcapacity) system failures. This traditional management approach utilises a system of restrictions and enforcing actions, through a top–down process launched by a central authority. In the technical literature, the system is categorised under Command-and-Control regimes (C&C regimes). Quota-based regimes are applied to diverse fisheries all over the world. They play, for example, a central role in the management of EU fisheries. In spite of its wide application, C&C quota-based regimes are often referred to as disappointing approaches to resource management. However, maintenance of the TAC system was widely supported by the industry during the last revision process of the CFP. This was in spite of the fact that, due to their design, C&C regimes leave little room for user participation and, consequently, do not include users’ knowledge as an input for decision-making. This weakens legitimacy and, consequently, generates non-compliant behaviour, such as misreporting and discarding. This undermines the quality of the data used to produce mandated scientific knowledge of the resource. As a result, managers are challenged to strengthen overall knowledge production through the use of all sources of knowledge and scientific disciplines.

6.2 PORTRAYING COMMAND-AND-CONTROL REGIMES

Scialabba (FAO Glossary) defines C&C regimes as instruments related to policy and management (mechanisms, laws, measures) that rely on prescribing rules and standards and use sanctions to enforce compliance with them. Resource management rules are prescribed by the regulator who usually leaves little flexibility for actors in implementation of the system (Alcamo et al., FAO Glossary). A characteristic of a C&C regime is that the ‘top–down’ management process leaves little room for users’ participation. FAO [1] states that the measures a C&C regime takes usually entail the setting of regulatory norms
and standards that forbid or allow certain outcomes. These norms and standards generally focus on constraining the exploitation of natural resources and regulating how a specific activity must be carried out within those constraints. In spite of that control, surveillance and sanctioning of infractions are important features of any management system, and are especially notable in a C&C output regime. These tools help managers to attempt closer control of users’ actions. Thus, control, surveillance and sanctioning of infractions are tools of a centralised decision-making system, in which the set-up of rules, and their close observance by the central authority, seeks to rule the exploitation of a given resource by limiting the catch (output regulations) and, in the case of a C&C input regime, the effort (input regulations).

C&C output schemes are widely used by governments, sometimes on request by the industry [1]. Pope [2] points out that output management is widely used and normally accepted by administrators because it is relatively easy to implement and monitor. In most fisheries in the world, fishery resources are public goods and the central authority rules their exploitation on behalf of citizens. C&C output regimes are especially useful when managing fisheries that cover large geographical areas. They have an obvious appeal when fishing opportunities have to be shared between countries, or communities or fleets, because it is easier for countries to agree on sharing catches in some proportion than to agree on sharing out fishing effort [1]. In a C&C output regime scenario, the centralised decision-making apparatus better suits the management of fisheries that are exploited under diverse jurisdictions and that require effective resource allocation. According to FAO [1], quota regimes are considered to be fair, up to a certain level, since the same rules apply to everybody. The output approach has also been demonstrated to be more suitable for fisheries based on few species [2].

It is worth emphasising that C&C output management is commonly perceived as a disappointing approach to management [1]. C&C output regimes encourage increases in individual efficiency due to the generation of ‘regulated’ open access in which each fisher tries to obtain a bigger share of the resource. Such competition generates redundancy of fishing capacity, which threatens the resources. In this scenario, property rights are weakly defined and the management system rigidly controls a TAC as a desirable level of resource exploitation. But the manager does not exert effective control over vessel participation [3]. As a consequence, access to the fishery remains free and open within the output constraints [4]. Due to the common pool nature of this management approach, individuals have an incentive to take a bigger share of the TAC. In this stage, the phenomenon known as the ‘race for fish’ will take place: in order to ensure their individual participation in the fishery, fishermen will invest in more fishing-empowered vessels. The rationale underpinning this is that fishermen will invest in new capacity as long as individual profits are larger than the costs. However, this will inevitably result in fishing capacity accumulation, which, due to the non-malleable nature of this type of capital, will continue beyond the level required to harvest the TAC. Due to the subsequent rent dissipation, the fishery will become vulnerable to adverse economic and resource shocks. In a situation of severe crisis, fishermen may press the government to provide subsidies in order to alleviate economic distress. Moreover, the redundant capacity will become a threat to other resources if participants push managers to increase the TAC or to divert effort to other fisheries, which is known as spill over effect [3]. To sum up, traditional
C&C output management brings the seeds of its own failure [2]. Furthermore, this failure often leads to the imposition of more restrictions [5]. The failure of C&C output regimes can be found in the very nature of non-participatory governance, in which the manager centralises decisions. The rigid decision-making framework does not provide space for user participation. This exclusion of users in a ‘top-down’ regime weakens legitimacy of the regime and, therefore, provokes non-compliance with regulations. Moreover, it encourages users to pursue merely individual objectives. The resultant lack of compromise in the process of resource management is another reason for the aforementioned ‘race for fish’ and redundant capacity generation.

6.3 MANAGEMENT OBJECTIVES

The primary objective of a C&C quota-based regime is to protect a given stock from over-fishing through the set up of a maximum allowable take to be exploited within safe limits. In Europe, every Member State (MS) receives a fixed share or quota from the TAC for a given stock. The diverse mechanisms to take and enforce the national quotas are decided and applied by each MS. Thus, national quotas are locally managed in different forms (IVQs, ITQs, etc.), according to the particular objectives of each MS.

Management objectives are not only focused on resource protection; other fisheries management goals comprise economic and social objectives. The setting of the TAC has to counterbalance objectives conflicting with the main objective of resource protection. For example, increasing short-term profit. This is a particular hard task since a small TAC may be detrimental to profitability in the short run, while favouring sustainability and, therefore, improving economic benefits in the long run [6]. High TACs may also sustain high levels of employment at the expense of sustainability. Conflicting objectives can also arise when a TAC is allocated among user groups. The hard task is to fairly allocate the resource, taking into consideration the needs of the users and their dependence on the resource. The problem is that, in C&C regimes, it is not only the central administrations who have objectives. User parties, which range from communities to countries, also have their own interests and disparate objectives. In a C&C regime, the rigid ‘top-down’ decision-making apparatus restricts managers to base their decisions mostly on their own objectives. This fact generates conflict with users before and after the allocation process. The objective-setting process gets more complex when other concerned groups as conservationists demand that managers take conservation objectives into account: for example, conservation of biodiversity in areas endangered by trawling.

6.4 MANAGEMENT TOOLS

C&C output management of fisheries resources is complex, as it tries to achieve resource protection and sustainability by taking measures that often clash with users’ objectives. These contrasting positions require managers to be aware of the different implications of their decisions in order to achieve sustainable exploitation. Two tools are involved in this approach to management: the TAC-setting and the TAC allocation.
6.4.1 The setting of the TAC

In order to set up a TAC, a fishery manager requires accurate information about the dynamics of the resources. However, in order to set an adequate TAC, it is not only biological factors that have to be taken into account. The manager needs to also take into consideration economic, social and political objectives and the trade-offs between them. Typically, the estimation of the TAC is made before the fishing season begins, based on information about the fishery in the past, harvest levels, and estimates of stock size, and may also reflect biological and economic modelling—for example, production functions [7]. In addition, other factors, such as political needs, are usually taken into account in the establishment of the TAC. It is important to highlight that those factors are so influential that, in many cases, they undermine reductions in the advised TAC. An example of the latter is the case of the advice for the Bay of Biscay anchovy fishery for 2005, whereby the Commission, following the advice of ICES, proposed to cut the TAC from 30,000 tonnes to 5,000 tonnes. However, the Council of Ministers at the December 2004 meeting decided not to reduce the TAC, with no clear arguments for their making that decision, other than social and political pressures.

EU fisheries are a good example of TAC-setting and allocation. Every year, the Council of Ministers meets in December in order to decide on the TACs for the following year. It is the end of a long process involving many countries and organisations. The International Council for the Exploration of the Sea (ICES), an inter-governmental body with 19 members, plays a key role in the advisory process. ICES is responsible for collating data on fish stocks collected by scientists from member states and associated states. Then, ICES’ Advisory Committee on Fishery Management (ACFM) reports on the status of over 100 fish and shellfish stocks, and on the consequences, in the long and short-term, of various limits on catches. The European Commission consults its own Scientific, Technical and Economic Committee for Fisheries (STECF), which is composed of national experts. Negotiations also take place with non-Community countries, which have an interest in the same fishing grounds or stocks, and with relevant regional fisheries organisations. The Commission then analyses the various opinions and sets out proposals for the following year’s TACs and the conditions for their harvesting. These proposals are sent to the Council of Ministers. The final decision regarding TACs and related measures are taken by the Fisheries Ministers at the December meeting. The TAC system offers a neat mechanism to allocate fisheries resources among the Member States. However, the use of the TAC system within the EU fisheries has major drawbacks. For political reasons, the TAC set by the Council of Ministers is often higher than the one proposed by the Commission, which follows the scientific advice of ICES [8].

6.4.2 The allocation of the TAC

Since allocation has to be done between countries, regions, communities, vessels or fishers, it is natural that conflict arises between the participants before and after an allocation. This conflict can be attributed to the conflicting objectives that concerned parties may hold. In the Community context, in order to share fishing opportunities among MSs, the Commission devised a formula to divide the TACs. This formula reflects the
Command-and-control quota-based regimes

The formula is based on what is known as the principle of 'relative stability', which ensures each MS a fixed percentage out of a given TAC (each species). This principle is based on the historical distribution of catches of each species between MSs, which supposedly reflects the socio-economic importance that those countries place on a particular resource.

However, it is necessary to question if the quota allocation provides a good fit for the countries’ dependency on the resources. Two examples can be given in which the quota share does not reflect this degree of dependency. The case of the northern hake fishery is a good example. Spain was allocated a quota that did not satisfy either the real catches or the local demand. Bilateral agreements between Spain and France tried to resolve the problem through quota-transferring. A second example is the anchovy fishery in the Bay of Biscay. In this case, Spain was assigned a quota far larger (90%) than the one allocated to France (10%), based on historical catches. However, the development of a brand new fleet—particularly pelagic pair trawlers—during the 1980s, immediately after the fixation of quotas for the anchovy fishery, resulted in an excess of capacity on the French side, alongside a scarcity of marketable resources. France tried to increase its quota share by means of bilateral agreements (the so-called treaty of Arcachon) and other more controversial means (cession of quota from the Portuguese anchovy population in Portugal, ICES Division IXa). Through the Arcachon treaty, Spain concedes part of its participation on the anchovy fishery to France in exchange for a participation in the exploitation of other species, including northern hake. These examples illustrate some of the problems surrounding relative stability. On the one hand, relative stability might not reflect real fishing practices at the setting up of quotas but particular conditions accepted by a Member State at the moment of joining the EU, as it is the case for Spain in the northern hake fishery. On the other hand, given that the Commission is responsible for conservation and management policy, while MSs are responsible for the development and activity of fleets, it might well happen that the two policies—conservation of resources and fleet development—are inconsistent and even conflicting, given rise to severe conflicts between national fleets and between MSs and the Commission.

After the TAC allocation has been agreed among MSs, it is the responsibility of each MS to monitor its quota uptake and to close the fisheries when quotas have been taken up. The MSs must keep the European Commission regularly informed about their quota uptake. This enables the Commission to monitor the situation at a Community level. Despite the fact that the TAC system is used to manage EU fisheries at a macro-level, quota management options vary across countries. Quotas can be kept in a national pool being allocated to producers’ organisations or even to individual vessel owners. In a few cases, quotas can be hired, bought or swapped—these are ITQs.

6.5 THE PRODUCTION OF KNOWLEDGE

In a C&C quota-based regime, knowledge is produced in response to government mandates. Governmental institutions create and validate knowledge, which is used as an input in decision-making. Since knowledge is usually produced from the interaction of government and research institutions, users have little or no participation in knowledge
production. Jentof [9] points that power plays an important role in knowledge production and validation. According to him, managers and scientists are in a powerful position to define the valid and relevant knowledge to be used in decision-making, while fishers do not. Such an exclusion from the consultative and decision-making process generates the reluctance of users to comply with regulations because it does not build a sense of ownership and, consequently, weakens legitimacy (Fig. 6.1).

It is worth highlighting that, within the circle of the C&C knowledge production system, political decisions sometimes ignore scientific advice developed on biological grounds. There is a paradox here. Administrators need a scientific apparatus to back their decisions and present them to the public. However, the ultimate decision of managers often ignores scientific advice. This decision is usually based on political aspects aiming to preserve users’ livelihoods. An ideal approach to modern management should be to take into account the expertise and knowledge of those who have a stake in the fishery. This is likely

![Diagram](image)

Fig. 6.1. This figure shows a simplified representation of knowledge production within a C&C quota-based system. The exclusion of fishers from the process of assessment and advice induces system failures, such as discarding and misreporting of catch data. This, in turn, undermines the quality of data used as inputs in stock assessments. The final result is flawed assessment and advice. (Source: Own elaboration.)
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to ensure equity and legitimate the processes of assessment, advice, decision-making and enforcement. Moreover, this is likely to incentivise compliance behaviour. As Wilson [10] sees it, the knowledge basis for management should be contributed to, shared and controlled by more stakeholders. This would help fisheries management to be more equitable.

6.5.1 Knowledge of biological and ecological factors

6.5.1.1 Biological knowledge

In a C&C output-based regime, the setting of an accurate TAC is the backbone of the system whose aim is to reach long-term biological sustainability. Setting an accurate TAC requires a comprehensive knowledge of the target stocks. The basic knowledge required comprises a variety of indicators such as yearly catches, which are normally compiled by national administrations and validated by research institutes. Abundance indices, such as CPUE of specific fleet segments, usually selected on the basis of gear and spatial range of activity, have also to be included in the data-set, and used as an input in production models. Most analytical assessments of North Atlantic stocks, for example, use length or age structured models (Virtual Population Analysis, VPA), which require that the length and age compositions of catches be estimated. Depending on the assumptions of the model to be used, it is necessary to estimate effective effort—for example, effort targeted at different species—or to standardise effort to remove the effect produced by changes in efficiency or in tactics overtime [11]. Regarding data from scientific surveys, indicators coming from trawl surveys, acoustic surveys and egg surveys are commonly used to provide input data for assessments. It is necessary to highlight that when scientific surveys are not carried out on research vessels they should apply a statistically valid protocol. Certain biological data are also needed for assessments. These data comprise growth and weight of species at age and maturity. The data-set should also include data processed by research institutes, as corrections for misreporting and discards data collected on fishing vessels by scientific observers [11]. When multispecies models are used, extensive collection of stomach data is undertaken with the aim of generating knowledge on predation rates among species.

Vessel surveys may deliver a picture of only a temporally specific status of the stock and may cover only a fraction of the fishable area, thus failing to capture the spatial diversity of the stocks. This can lead to possible biased estimates of the stock, depending on stock structure and migratory patterns [12]. On the other hand, although detailed information on the amount and composition of commercial catches is crucial for the quality of stock assessment, catch reports may have substantial errors (the amount of fish landed is known but the current amount of fish caught is unknown), and this can seriously affect the reliability of the scientific advice regarding the future development of stocks and catches [13]. The problem is that TACs regulates total landings instead of catches [14]. This fact gives room to practices such as discarding and misreporting. These practices, in turn, undermine the quality of catch statistics (Fig. 6.1). Even though discarding practices are found in other management regimes they are exacerbated in a ‘top–down’ output regime where authorities are usually ‘far away’ from the fishers.
and their activities. Moreover, in a C&C output regime the TAC authorities seek to
rule landings instead of catches. This fact gives rise to activities at sea, such as high
grading, that occur when fishers aim to carry on board only highly valuable species,
when they prefer to use quotas when prices are higher, or when they dump undersized
fish to avoid punishment. Regarding misreporting or underreporting, these problems are
also encountered in other regimes, but are especially common in C&C output regimes.
The reasons for this behaviour go beyond purely economic reasons, such as avoidance
of taxes or quota exhaustion: there is a meaningful component of a lack of compromise
with the management system, which is generated in the minor or negligible participation
of fishers in the management process. This creates an incentive for the fishermen to
pursue individual interests to the detriment of general interests, such as stock protection.
This lack of compromise is also a component of discarding. These factors undermine
the quality of input data for assessment. Biologists as well as social scientists should be
aware of these problems and develop methodologies to estimate the extent of discarding
and misreporting in order to include them as a component in assessment.

6.5.1.2 Ecological knowledge

According to Cochrane [15], the inclusion of ecological data in fisheries management
is recognised to be essential to sustainable and efficient use of resources. Since target
species are dependent for their survival on the productivity of the ecosystem in which
they live, any change in that productivity will impact target resources. The manager
needs to be aware of such changes, whether natural or caused by fishing or other human
activities: for example, the impact of trawlers on the seabed or the impact of discards.
Such an awareness would enable assessment of the possible impact of ecological changes
and also the adoption of management measures to minimise damage to the ecosystem.
Among the data required to provide information to fisheries managers are: by-catch
estimates by species or indicator species and annual discards estimates per fleet; length
and age composition of by-catch species or indicator species; and the impact of fishing
gear and activities on the physical habitat and changes in habitats caused by other human
activities.

Regarding discarding, this issue is one of the most acute problems within a C&C
output regime, since it not only undermines the quality of data but also generates a
negative ecological impact. For instance, sea birds, mammals or other organisms can
become dependant on discarded fish. The challenge to the scientists here is to conduct an
accurate estimate of discards and to measure the magnitude of their ecological impact.
Stakeholders groups, such as conservationists, have a say in these issues and, in fact,
have raised their voices to prevent impacts of certain fishing practices (e.g., mammals
by catch, ghost fishing, etc.) on the ecosystem. Despite the fact that their objectives
commonly clash with other groups’ interests, which generates conflict, the inclusion
of conservationists’ knowledge is one of the great improvements in modern fisheries
management. Thus, opting for a participatory approach and, therefore, moving away from
a C&C management system offers the potential to alleviate ecological problems caused
by fishing.
6.5.2 Knowledge of social and economic factors

6.5.2.1 Knowledge of social and political factors

In a typical C&C quota-based regime, social factors are not usually included as inputs in assessments and the advice derived from this. However, political and social factors are taken into consideration in the decision-making process. These factors often guide manager’s decisions. Frequently, these factors are the sole basis for decisions such as TAC-setting. It is worth emphasising that managers establishing TACs solely based on the aforementioned factors should remember that the prime objective of management is to exploit the resource in a way that ensures the sustainability of their exploitation.

The challenge to scientists is to develop a solid methodology that uses social and political science, together with biology and economics, to develop a framework to evaluate management measures. Knowledge about political and social factors should comprise a comprehensive understanding about the characteristics and degree of dependence that fisher organisations, industrial groups, communities and consumers place on the resource. This knowledge would allow advisers to make a prognosis about the behaviour of the various actors under possible candidate management scenarios. It would also help managers to take decisions that attempt to approach the various needs of the diverse actors concerned.

Managers should also hold information about levels of employment within the industry—for instance, the number of workers devoted to harvesting, processing and trading activities. It would also be useful to hold information regarding employment level in such fishing-related activities as ship-building and maintenance, netting material production and gear construction, etc. The collection of such information enables managers to measure the collateral impact of decisions such as closure of fisheries. Moreover, managers should hold information about labour opportunities outside the fishing industry. This is key information when taking hard decisions, such as closure of fisheries or vessel decommissioning. In this scenario, labour opportunities outside the industry would act as a safety valve for unemployment in the sector. To sum up, managers should be well-informed regarding the degree of dependency of the fishermen and community on the resources to be managed.

In a C&C regime, misbehaviour of fishermen generates several problems in data collection. In order to cope with these problems, the analysis of fishermen’s behaviour is another of the tools to be used in achieving an effective management of resources. It is argued that negative forms of fishermen behaviour, such as misreporting of landings, discarding or non-compliance, undermine the quality of the data collected. This generates negative consequences, such as flawed assessments that tend to distort the perception of the stock status, undermining the credibility of the system and potentially driving over-exploitation. Key factors such as discarding require analysis and understanding from the point of view of socio-economics in order to be included in models that allow the evaluation of candidate management systems [14].

Steelman and Wallace [16] consider that biological and economical principles, and social behavioural theory, need to be integrated in the design of management policy. In particular, social behaviour considerations, such as self-interests, norms, rules and expectations, as well as the history and culture of the community, can influence the sustainable
utilisation of fisheries resources. Despite the weakness of C&C quota regimes, Steelman and Wallace [16] consider that their degree of success or failure depends on specific social factors. In particular, they affirm that when there is no cultural connection to the resource or history of stewardship and where regulatory strategies do not conflict with the desires of the affected population, a C&C regime may be a good option for management.

6.5.2.2 Knowledge of economic factors

In a C&C output management regime, managers should have a comprehensive knowledge of the underlying economic factors affecting fishermen’s and the fleet’s behaviour. Fortunately, knowledge production regarding the performance of this system is meaningful and shows, by theoretical bio-economic modelling and empirical application of the models, the behaviour of the fishery under a C&C output regime. Clark [17, 18], Pearse [19], Arnason [20] and Christy [21] inter alia, have criticised C&C as a biological management regime that introduces economic inefficiency, the main characteristics of which are overexploitation and fleet redundancy. Several examples support theoretical findings regarding the failure of the C&C quota regime over the long run. For example, it was observed in the Pacific halibut fishery shared by the United States and Canada [22]. This fishery was facing a heavy depletion of the resource. The Pacific Halibut Commission managed to rebuild and maintain the resource. However, it failed in controlling the fleet size, which constituted a threat to the resources and generated economic inefficiency.

When managing under C&C regimes, managers should be provided with a comprehensive set of economic information. This data set should comprise average income and costs per fleet on a yearly basis (if possible disaggregated by fleet segment), profitability of the fishing fleets, destination of landings, and prices of fish [15]. It should also comprise data on capital investment, capital productivity, rates of return, and gross and net value-added. Regarding indicators, it is quite important to consider the discount rates. The choice of an appropriate discount rate has much more influence on any retrospective analysis, forecast or outlook related to sustainability than any other factor. A fishery system generates a cash-flow to both current and future generations. Since a meaningful uncertainty exists, risk management policy must be ‘flexible’ (rather than ‘static’). Thus, managers should consider integrating option pricing and contingent valuation models as they develop models consistent with the existing uncertainty. Those models also offer the options to compute the opportunity cost, which is another additional indicator to include as part of the management plan. Moreover, contingent valuation methodology could also be considered, since coastal resources generate not only economic cash-flow, but also economic goods without a market price.

Fishing capacity evolution also needs to be analysed. Thus, holding comprehensive information about the current fishing capacity and, when possible, economic and technical efficiency of the fleets, will help to keep track of fleet evolution. Assessment of fleet efficiency can be conducted by the application of production theory tools, such as Data Envelopment Analysis (DEA) and Stochastic Production Frontiers (SPF), which require comprehensive information on inputs: for example, fleet fixed and variable costs; and vessel characteristics such as age, dimensions, engine power, gross tonnage, etc. Holding relevant information about the fleet will help managers to understand how fishing capacity
threatens the target species. Moreover, it will help to make a prognosis of the potential spill over effects to emerge and, thus, if there is the need to impose restrictions on the harvesting of target stocks. Because of their intrinsic characteristics, C&C output regimes demand robust monitoring, control and surveillance (MCS) and enforcement systems. In this context, managers should be aware of the costs of implementing a C&C regime in a given fishery. When fisheries are too large (for example, fisheries that cover a large number of vessels, fishers, geographical areas, and a big number of marketing channels), both enforcement and monitoring become an extremely difficult task because of high costs [16].

6.5.2.3 Knowledge of technological factors

Management regimes operating under C&C systems (whether catch limit or effort limit) are affected by technological creep, which means that fleet technical efficiency increases due to technological improvements. Examples of technological creep are the incorporation of new or modernised fishing vessels equipped with sophisticated fish-finding devices, state-of-the-art deck equipment, global positioning systems (GPS), and new communications technologies. Improved fleets are progressively more able to find fish schools regardless of stock abundance. In these circumstances, abundance indexes such as catch per unit of effort (CPUE) are less reliable because well-equipped vessels operated by experienced crews are able to find fish schools and effectively catch them. This phenomenon can be mostly observed in pelagic resource exploitation [23].

Keeping an updated database of fleets that comprises the main technical characteristics, fleet structure, catches, etc. will allow managers to assess the evolution of capacity and, if possible, of technical efficiency. It will also allow them to conduct a prognosis regarding what possible impacts measures such as decommissioning schemes and renewal of fishing vessels would have on fleet capacity and fishing effort. Knowledge of technical efficiency is particularly useful when carrying out decommissioning schemes because it enables managers to select which boats could leave the fishery. For example, meaningful reductions in capacity might be obtained by withdrawing the most efficient vessels. The development of a comprehensive register of fishing vessels should not only comprise technical details of the vessel, but also details of ownership and records of violations. This register would back up MCS labours.

It is desirable to be aware of technological improvements and to keep these in mind when facing problems generated by fishing practices. This is the case for by-catch excluding devices and selectivity devices. Managers should also hold knowledge of the impact of certain gears on the environment, particularly impacts on the seabed, and encourage fishermen to apply environmentally friendly netting materials and gear designs.

6.6 LEARNING FROM INNOVATIVE SYSTEMS: TAKING INTO ACCOUNT USERS’ KNOWLEDGE

It is not surprising that users’ knowledge is not taken in account in a traditional and fairly pure C&C output regime; although it is advisable to incorporate this kind of knowledge in order to bring a stream of experiences and information to improve all the
fig. 6.2. Process suggested for assessment, advice and decision-making in fisheries management. Note that the inclusion of fishermen and industry tries to strengthen legitimacy in the management process and avoid data distortion (note the contrast with fig. 6.1). The process suggested includes traditional knowledge and multidisciplinary scientific knowledge as a basis for decision-making. (Source: Own elaboration)

processes of management (Fig. 6.2). Charles [12] sees the advantages of one kind of users’ knowledge: traditional ecological knowledge (TEK). For Charles [12], TEK is one of the key elements to be included in management in order to enhance its effectiveness. Resource users and coastal communities hold wisdom about nature and about suitable management arrangements for their cultural and belief systems. They can also assess workable approaches to improve compliance among users, and discern which fisheries techniques are more effective and sustainable within the local context. Thus, it is advisable to take TEK into consideration when deciding how to allocate and subdivide a given TAC in a given region—for example, by community quotas, fishing gear quotas, vessel quotas, etc. In some modern fisheries systems, the decision to set a given global quota involves different users through hearings. A good example of users’ participation is the case of the Canadian Atlantic groundfish fishery. According to Charles [12], the Federal Government has made efforts to modernise the Management Act. A co-management approach has been introduced, in which communities, industry and government work together to develop and enforce regulations. The management system is based on a TAC, which is agreed after a process in which scientists have an informative role. This information is used together with input from public hearings, in which any stakeholder has the right to participate. Then, this information is presented in public to the Minister, who decides the TAC. The global quota is divided into shares, allocated to each sector
in terms of location, gear type and vessel size. Each sector is responsible for presenting its own Conservation Harvesting Plan (CHP) in which it is stated how the sector is to fish within allowable limits. The CHP have to meet official conservation requirements before fishing is allowed. The CHP details allowable fishing gear, at-sea and dock-side catch monitoring, and measures for closing the fishery in case of undersized fish catch incidence.

User participation is also found in the USA, where a system characterised by the open nature of the management process is found. This institutional change took place in 1976 with the implementation of the Magnusson Fishery Conservation and Management Act [24]. The Act instituted management councils in which a wide variety of interested parties are allowed to participate (including state and federal agencies, fishermen, environmentalist, researchers, consumers, processors and the public). The responsibility of the councils is to design management plans for fishing activity within their respective territories. However, plans are sanctioned by the federal government prior to implementation. During the planning process, important input is taken from public hearings [25]. Thomson [26] refers to a good example of user participation in the USA—the Pacific salmon fishery—in which input from users, together with research (surveys and computing modelling), is used to build harvest rules. According to the Washington Department of Fisheries and Wildlife (WDFW) [27], participation of users is through deliberations that involve representatives of the industry, treaty tribes, several states, Government and the public. Despite the fact that these fisheries somehow still utilise a ‘top-down’ system, they are getting closer and closer to user participation: not only in utilisation of users’ knowledge; but also in decision-making.

6.6.1 The DPSIR framework

During the 1980s and 1990s, sustainability concepts originally focussed on resource and environment protection and were developed towards models that take into account socio-economic, community and institutional concerns. These models, in general, aim for the well-being of people; especially for those within small coastal communities. Sustainability, with its corollary of resource conservation, becomes the means to achieve the long-term welfare and preserve the livelihoods of coastal communities and the stability, integrity and functionality of their social and cultural fibre. In order to coherently organise those interactive issues, or domains, pertaining to the sustainable development of a system, a framework is necessary and must be developed. This framework should show the various dimensions of the system, the factors at work and their relationships, as well as the key criteria for which indicators need to be established and monitored. There are a number of potential frameworks to be considered: (i) the Code of Conduct for Responsible Fisheries; (ii) the FAO definition of sustainability; (iii) the DPSIR framework. The DPSIR model developed by the Organisation for Economic Cooperation and Development (OECD) is an expansion of the PSR (Pressure-State-Response framework) in which it is considered that ‘driving forces’ more accurately reflect the economic, social and institutional dimensions of sustainable development. In the expanded framework human Driving forces (demand for food and revenues fuelled by economic and demographic forces) exert Pressure that results in changes in the State of the components of the system and its environment.
(decrease in resource biomass or in revenues to coastal communities), and may have an immediate Impact on the functioning of the system (collapse of fisheries, social unrest, decline in compliance). Societies, through their management authorities, provide a Response to these changes in state and their effects (legal, institutional and or financial measures) with a view to modify the pressure (through management) or to mitigate its effects (insurance schemes, contingent plans, etc.) [28].

The DPSIR model has limitations with regard to both miming how the world works and guiding policy decision-making. This is because it is a linear cause-effect model, and, as such, it oversimplifies reality and ignores many of the linkages between issues and feedbacks within the socio-ecological system. This simplicity is an advantage for those willing to take action; but it is also an easy argument against using it for those reluctant to do so. A possible response to such criticism would be to use the DPSIR framework but integrate linkages through other tools such as modelling (bio-economic models are the best example). In any case, the DPSIR framework is considered by many as a practical starting point for the organisation and presentation of indicators; although, in practice, it is not always simple to distinguish between indicators of ‘state’ and indicators of ‘pressure’.

6.7 INSTITUTIONAL SUPPORT FOR KNOWLEDGE PRODUCTION, ADVICE AND DECISION-MAKING

Scott defines institutions as cognitive, normative and regulative structures and activities that provide stability and meaning to social behaviour [9]. Institutions have an influence on knowledge production and validation. In modern fisheries management and especially in C&C regimes, scientific knowledge is produced in response to mandates and used as an input in decision-making. The problem is that, since knowledge is produced within a rigid administrative institutional framework, it is normally produced by specialised institutions, which gives little room to informal knowledge held by the fishers and industry. Due to the prestige of scientific knowledge, policy makers aim to back their decisions with science [29]. Moreover, high expectations on the performance of research institutions are held by managers, stakeholders and the public. In this context, knowledge by state research institutions becomes the main form of knowledge production. The outcome of knowledge production is not only expected to be accepted and used by management institutions, it is also expected to be accepted by stakeholders and fishers. Non-compliant behaviour can be expected from users who do not believe research institutions are conducting accurate assessments of the state of resources or delivering accurate recommendations. Thus, a lack of trust in the research institution will add a component of uncertainty, which can generate, among users, a wish to exploit the resource at a higher rate. This fact is especially noticeable in C&C quota-based regimes, in which knowledge production and decision-making are commonly taken without considering stakeholders perspectives.

Regarding decision-making, Morgan [13] points out that in countries where quota management has been introduced, it is a government body that usually determines quotas and administers and enforces them. Thus, it is clear that there is a correlation between the failure of quota-based regimes in reaching their objectives and ‘top-down’ decision-making. Possibly, this failure can be attributed to poor institutional arrangements that
have divorced governments and industry functions. A lack of understanding of industry needs by scientists and managers and a lack of trust by the industry regarding the process of setting and allocating a TAC can drive a C&C regime to failure. In contrast, joint efforts by the Government and industry are key factors of success. Innovative approaches that include industry participation in management are carried out, inter alia, in Canada, the USA and Australia.

6.8 CONTROL, ENFORCEMENT, AND COMPLIANCE

Due to their intrinsic problems, C&C output regimes require a robust control and enforcement apparatus to back their functions. This is because the central authority requires control and enforcement to pursue the goal of resource protection. Since discarding and misreporting of landings are some of the negative outcomes of the system, counteracting these problems becomes an expensive aspect of the management process. Carrying out a system of MCS not only demands a high budget, but it also requires skilled personnel, training and infrastructure. The knowledge that should back MCS comprises information about fishing practices, fleets, possible infringements and expertise in the use of MCS infrastructure. A strong enforcement system should reduce non-compliance. However, it will increase costs and reduce flexibility. Moreover, it will incentivise the circumvention of regulations, which will, in turn, generate more restrictions. A clever approach to enhance compliance would be to take into account fishers’ knowledge about what arrangements regarding control and enforcement will work best in their communities.

Understanding the mechanisms that lead to non-compliant behaviour requires knowledge of fishers’ socio-economic motivations. Non-compliance not only depends on fishers’ expected gains from illegal practices, but also on social aspects. These aspects are embedded in a local context in which, inter alia, enforcement, participation and moral factors plays a major role. Nielsen and Mathiasen [30] see some key factors that have a major impact on compliance:

- **Economic gains to be obtained from non-compliance and the risk of being detected and the severity of the punishment.** Even though, this first incentive mentioned by the authors is difficult to deter, a system of strong enforcement can deter the incentive to cheat. However, it should be backed by a strong and effective judicial power.
- **The compatibility of the regulations and the practices of fishing.** Usually, the setting of technical regulations is taken by the central authority, backed by scientific advice. However, the issuing of technical regulations, especially those related to harvesting, are often not in accord with fishing practices. Thus, using fishers’ knowledge as an input in the set-up of regulations will avoid the problem of introducing un-workable measures.
- **The perception that the regulations conserve fish stocks.** If fishermen have the perception that regulations do not conserve the stocks, it will create uncertainty. Therefore, it is likely that fishermen will be eager to take as much fish as possible, as fast as possible, in many cases by breaking the rules.
The actual and expected behaviour of other fishermen and the morals of each individual. There is little the management system can do to enhance moral behaviour of fishermen. However, educational programmes may help fishers to understand that non-compliant behaviour affect colleagues more than managers.

Management alternatives, such as co-management, have a positive impact on compliance. As it is widely understood, the incorporation of user groups’ knowledge and active participation in the management process brings about legitimacy: a fact that enhances compliance.

Enhancing compliance within a C&C quota-based regime is always difficult due to the fact that TACs are commonly specified at a certain level of landings rather than at a certain level of catches. Due to this system failure, fishers are incentivised to misreport and discard fish or to commit illegal transhipment at sea. Moreover, since the MCS system mostly focuses on output, fishers have the opportunity to increase inputs; sometimes illegally. A sensitive approach to management in a quota-based scenario should be to enhance compliance instead of only developing restrictions. Pope [2] suggests that backing enforcement with an educational programme would make fishers understand that non-compliant behaviour affects other fishers rather than government.

Nielsen and Mathiassen [30], analysing the current TAC system within the EU, consider non-compliant behaviour as one of the reasons why the current system is not working properly; the other key cause being uncertainty in the stock assessment process and, thus, in the setting of the TAC. According to the authors, the lack of involvement of fishers weakens the legitimacy of the management system, which, in turn, may weaken incentives to comply with regulations. They finally suggest that, to enhance incentives for compliance, the management process needs to involve fishers too. Thus, everything indicates that a good means to alleviate the problem of non-compliance is the use of fishers’ knowledge or ‘bottom-up’ knowledge in the construction of the knowledge base for management. Within EU management systems, the implementation of the Regional Advisory Councils (RACs) is a remarkable development to bring users’ knowledge, and their inclusion and participation, into the management process.

6.9 CONCLUSIONS

Sustainable management of fishing resources is a goal generally aimed for by all fishing states. The problem is that effective fisheries management not only faces intense environmental change, but also important impacts from biological, socio-economic, political and technological factors. The synergies between all these components are so important that many argue that there exists just one integrated fishery system to be managed. In a fishery system, sustainability has to be considered at various levels: the resource and its environment; the social and economic needs; and the technological component. Yet it is difficult to take into account all of these inter-related domains simultaneously.

Fisheries management has traditionally focused on issuing institutional measures oriented to resource-protection. C&C output-based regimes represent a rather appropriate
management tool to achieve this goal. However, the rigid C&C ‘top-down’ decision-making process is backed by biologically oriented management research and does not take into account users’ participation and socio-economic knowledge. Furthermore, a C&C output regime confines managers to basing their decisions mostly on their own objectives. Thus, for scaling up into a more integrated management quota-based regime, interdisciplinary knowledge is required as an input to the management process.

Initially, there are two types of information required: biological indicators, such as resource abundance, catches, growth, weight at age and maturity; and ecological information, such as interrelations between species and impacts of fishing activities on the ecosystem. This second type of information will require the integration of social, political and economic information. However, frequently, social and political factors are solely used to take decisions such as TAC-setting. Knowledge about both factors could provide essential information about the characteristics and degree of dependence that fisher organisations, industrial groups, communities, conservationist groups and consumers have on the resource. In general, managers must try to obtain and include information *inter alia* about the levels of employment within the industry and fishing-related activities, the structural inflexibility of families and communities, the degree of dependency on fishing, and fishermen’s behaviour.

A C&C quota-based management regime also requires the integration of economic indicators, such as fishery net revenues, capital and operating costs, profitability of the fleets, opportunity costs, and levels of investment in the fleet. A particular contribution by economic knowledge could be the incorporation of specific information about the efficiency and capacity of the fleet. Recent years have witnessed the efforts of science to integrate technological knowledge. An integrated C&C management regime must also consider *technological creep*, evolution of capacity and the impact of fishing technology on the ecosystem.

This interdisciplinary process of integrating, interpreting and communicating knowledge from several scientific disciplines must be complemented with users’ knowledge. Effective management and sustainable resource development will be benefited by an expert-based approach to management, which enables the incorporation of users’ expertise. Institutions close the list of factors to be integrated within the management process. Among the institutional structures to back up management are the control and enforcement system. In particular, C&C output regimes require a robust control and enforcement apparatus, since illegal activities such as discarding and misreporting of landings are some of the negative outcomes of the system.

Currently, new tools are being developed to improve management. This is the case of the Driver Pressure State Response (DPSIR) framework developed by the Organisation for Economic Cooperation and Development (OECD). The framework is a useful device to clarify the inter-relations between interdisciplinary factors and to highlight the characteristics of ecosystem and socio-economic change. An ideal combination of factors/indicators could be fed into this conceptual model which would identify what, how, and why change is occurring within such a complex system. It is worth highlighting that, in the EU, an important and transcendental innovation is taking place. This is the creation and implementation of the Regional Advisory Councils (RACs), which have
been created as a means to improve management through the incorporation of users into the management process.

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REFERENCES

Command-and-control quota-based regimes


Effort and Capacity-Based Fisheries Management

7.1 INTRODUCTION

Effort based management is an alternative management system to the total allowable catch (TAC) system. The European Union (EU) has started to use both management systems in parallel by supplementing the traditional TAC fleet capacity restrictions with sea-day restrictions from 2003 (see Section 7.7). However, global examples of effort-based management systems are scarce. In this chapter, the knowledge base of command and control effort and capacity regulation is described and evaluated. This is based on literature review, as well as selected examples of use of different types of this regulatory system from a global perspective.

The knowledge base of effort regulation: The knowledge basis includes the scientific advisory needs and socio-economic institutional implications for different types of effort regulation. In this context, the adaptations of the management strategies, defined as the overall set and sum of management measures and instruments put in place by the management authority to regulate the fishery for achieving the management objectives [1], are also a part of the knowledge base, alongside the fishermen reactions (incentives, legitimacy concept, and compliance) to the management instruments. Different fisheries systems demand particular adaptations in the production of knowledge for management. Accordingly, the knowledge base comprise institutional structures, management procedures, data sources, analysis of stock and fleet dynamics, scientific advice and the communication of the advice under effort regulation.

Political choices in relation to the knowledge base of effort regulation: If fisheries management is to work, management objectives must be specified in order to judge success or failure of management. A range of political choices need to be made regarding the structure and contents of the knowledge base when implementing different types of capacity and effort management, as illustrated through all the examples of effort
and capacity regulation presented in this chapter. The specific management objectives being formulated can include: biological fisheries objectives (for example, single species biomass or fishing mortality rate reference points, Maximum Sustainable Yield (MSY) for target species, rebuilding over-exploited stocks or stock recovery, maximising protein supply, increasing selectivity and reducing discarding); broader ecological objectives (for example, conservation of resources from a multi-species perspective considering by-catch and the fisheries’ influence on overall environment, prevention of ecosystem shifts, protection of habitats, and protection of rare and sensitive by-catch species); economic objectives (for example, Maximum Economic Yield (MEY), maximising profit or resource rent, minimising variability in income, keeping prices low, raising government revenue, improving catch quality, and increasing exports); and social-political objectives (for example, maximum employment, maximising protein supply, reducing over-capacity, preserving the status quo, reducing conflicts and avoiding upsetting stakeholders, and boosting recreational fisheries). From such objectives, different management strategies, and how to apply those in terms of management actions, can be formulated, including allocation of fishing licenses—access rights to the resources—and individual effort quota between fleets (for example, the Australian case in Section 7.4). A problem is, however, that a management strategy can meet some objectives but conflict with others, and multiple objectives accordingly have to be balanced and trade-offs compared. Thus, common units should be developed to compare the effects of strategies in relation to the weighting of different objectives.

In this chapter the knowledge base of management and advice for implementing effort regulation, in partly national fisheries systems and partly multi-jurisdictional systems with the relative stability for historical rights of international resource allocation and sharing, are evaluated (for example, the North Sea and Mediterranean examples in comparison with Australian, Faeroese and Danish examples in Sections 7.3–7.7). Knowledge about fishers’ behaviour in relation to regulations is also addressed (Section 7.3), and economic incentives for effort allocation, switching between fisheries, discarding, and fleet adaptation are discussed. Overall knowledge of compliance and incentives to comply with regulations and the concept of legitimacy are comparatively addressed for effort management and TAC-based management among the selected examples (see Sections 7.3, 7.4 and 7.8).

7.2 THE KNOWLEDGE BASE OF EFFORT-BASED FISHERIES REGULATION

**Knowledge base of direct effort regulation:** Direct effort regulation is an input management system which requires a clear and precise definition of effort and capacity as a part of the knowledge base. According to FAO [1], input controls are restrictions put on the intensity of fishing activities and equipment used for fishing. That is, the number and size of fishing vessels allowed (fishing capacity controls), the amount of time the vessels are allowed to operate (vessel usage and activity controls), or the product of capacity and activity (fishing effort controls).

Direct effort regulation can be categorised into two types: (a) restrictions on overall capacity or restrictions of access for a certain capacity to given resources in given area and
Effort and capacity-based fisheries management

...season; and (b) restrictions on activity levels for certain parts or all of the total capacity. When evaluating the knowledge base for capacity and effort regulation it is important to consider the type of input regulation.

Effort management is based on fleet units, rather than on stock units as in a TAC system. Effort regulation should cover the full fleet and be based on homogenous vessel-categories (e.g., category vessel types by main gear such as trawlers as well as vessel length or engine power categories) where each are regulated according to their capacity, activity, and fishing pattern. Categorization of fishing vessels as well as tracking of fleet structure developments, fishing power evolution and technology utilization are central elements in the knowledge base for managing capacity and fishing activity. The restrictions and constraints of direct effort regulation include a license system with fishery and catch permits to allow fishing in certain areas and seasons and on certain species for particular vessels and gears, as well as restrictions on the capacity of licensed vessels (vessel size and engine power or gear size restrictions), which are often accompanied by the specification of maximum activity levels for particular licensed fleets. Licensed fishing rights might be transferable between owners and acquire substantial value (as in most of the examples).

Knowledge base of indirect effort limitations: Direct effort regulation is typically accompanied with and linked to technical regulations and management actions, which also include indirect methods of effort limitation. Examples of such methods are closed areas and seasons, which indirectly restrict fishery input, gear restrictions such as mesh size, and minimum landing size regulations. The technical measures control the catch that can be achieved from a given effort. Technical regulations are important in all the presented examples of effort regulation in this chapter. Additionally, by-catch and discard regulations are often connected to effort regulation systems; TAC restrictions may also cause limitations in effort or redirection of effort; and, finally, the possible transfer of individual effort quotas needs to be regulated to adjust overall fleet structure and fishing patterns (see, for example, the Australian example in Section 7.4).

It is necessary that the knowledge base contains information of types and aspects of different management actions to enable comparison and evaluation of different effort management systems in relation to management objectives.

Scientific advice production to support effort regulations: It is necessary to know how the fishery, stocks and society respond to different management actions based on different management objectives. Effort regulation systems depend on high quality effort and catch data, as well as sufficient information about discarding levels. A major aspect and problem of effort management is in defining a reasonable unit measure of fishing effort. This is because a major requirement of any input control is that a given amount of the input regulated should correspond to a constant ability to exploit fish [2]. Thus, to be able to evaluate management efficiency and to produce advice on the regulatory system, a precise definition of effort and high quality effort and catch information is necessary. Quantitative advice is concerned with calculating the probability that a given outcome will result from a given management option.

For those systems using age-based population dynamic models, and, for example, converting TAC to Total Allowable Effort (TAE) (for example, the Faeroese and North Sea examples in Sections 7.3 and 7.7), emphasis will be on fishing mortality rather than on catch quota. Such systems are based on estimation of the correlation between partial
fishing mortality and effort on a fleet basis. Such an estimate needs to be part of the knowledge base of the management system, and should account for the nature of mixed species fisheries and technical interactions. However, it can be problematic to model the link between fishing mortality and fishing effort. This is partly due to efficiency development over time ("technical creeping"). The TAE is established from the average historical correlation between fishing mortality and effort on a fleet basis. Partial fishing mortality of a fleet is estimated from the overall fishing mortality by its share of the total catch.

Effort-based management is less dependent on precise yearly stock assessment estimates (for example, Faeroese example in Section 7.3), which can instead be made on a multi-annual basis. This is in contrast to TAC-based regulatory systems, which do depend on precise yearly assessment and forecast estimates, and which often suffer from the problem of poor quality catch and discard data for recent years. If these estimates are inaccurate then the catch restriction might be set higher than the size of the stock, thereby endangering that stock and, consequently, resulting in no actual restriction to the fishery. In contrast, while effort management will also need reasonably precise stock assessments and forecasts to set an appropriate initial effort level and to respond to changes in efficiency, the removal rate generated by current fishing effort levels may be judged over a series of years rather than on a year by year basis, giving the advantage of averaging out extreme estimates [2].

In multi-species fisheries, effort switching between available species, depending on which is currently most profitable, has the risk of directed over-exploitation of the more valuable fish. However, effort management of mixed fisheries gives the ability to direct effort towards the most abundant species and to reduce fishing pressure on depleted stocks, albeit that this is dependent on stock distribution patterns (for example, the North Sea case detailed in Section 7.7 and Appendix 7.A).

To maintain relative stability in multi-jurisdictional fishery systems (for example in the EU, in Section 7.7), it will be necessary to convert the overall TAC or catch to effort by fleet according to partial shares of the catch by species. This will give conflicting effort levels by species in multi-species fisheries and, consequently, some stocks will be at risk of being over-fished and some under-fished, as only one stock can be fished on the single species optimal level. These problems of effort management and the associated knowledge base needed are discussed through the different examples.

Over-fishing of one stock in a given year in a mixed fishery can be compensated for in the following years with multi-annual effort-quotas between years. However, this demands stocks to be stable and sustainable. Stocks below a carrying capacity that fluctuates are sensible to even small levels of over-exploitation and are, thus, at risk of collapse. In many fishery systems, there has been a development from the stable exploitation of many year classes of one stock to a situation where mixed fisheries switch between target species of fluctuating stocks consisting of a few year classes of young fish (in, for example, the North Sea, Section 7.7).

Bio-economic assessment of fisheries is also increasingly quantitative and can provide a basis for meeting specified management objectives (for example, the Australian Prawn fisheries in Section 7.4 and the Danish mussel fisheries in Section 7.5), and can be used in the evaluation and prediction of overall performance of different management options and actions.
In assessing the impact of the fishery on the environment and ecosystem processes (ecological assessment), the focus is more often on describing the impact and its level as well as on formulating general and global ecological management objectives on the preservation of overall sustainability, rather than on formulating specific objectives and making exact quantitative predictions of the influence of different management actions on given factors (Australian example in Section 7.4). A key problem lies in how to define appropriate quantitative management indicators and measures of performance. However, more focus is now put on formulating direct management objectives and developing advisory models to quantitatively evaluate the impact of discarding and higher selectivity in fisheries (see Section 7.4).

**Performance utility criteria to evaluate effort regulation:** In order to determine whether management objectives are met, managers will need indicators. Biological and economic performance and utility criteria in relation to management objectives, and their sensitivity for the effort and capacity regulation, have to be defined and included in the knowledge base to evaluate the effect of the regulation. Quantitative criteria are easier to use in judgment of success of management rather than the more qualitative criteria.

Important biological and ecological criteria should relate to long-term sustainability and abundance of target and by-catch species, formulated through biomass and mortality reference points: stability of spawning stock biomass as well as catch and discard levels (fishing mortality) under effort regulation. Knowledge about changes in catchability, growth and stock production is important here. Other criteria could be the proportion of habitat impacted by a fishery, proportion of by-catch and discards, or species diversity.

Important economic criteria include stability in fisheries and fleet levels, optimal effort levels and effort displacement (between fisheries and areas), vessel profit in relation to investments (profitability), resource rent, revenue and costs, economic capacity, as well as aspects of the socio-economic efficiency of the command and control effort regulation system. The rate of change and variability in these criteria are important indicators. Social criteria cover, among others, levels and variability in employment in the fishery and number of vessels and capacity in the fishery. Other performance criteria are the concept of legitimacy of regulations and compliance to those—acceptance of the management system—where the number of illegal practices, conflicts and complaints are indicators.

These performance criteria must be a part of the knowledge base to aid in the evaluation of advice and management efficiency in obtaining an economical yield of the fish stocks and in understanding how the resources can be exploited optimally and sustainably, and to ensure exploitation works on market-based framework conditions under effort regulation.

**Incentives, legitimacy, acceptance and compliance in relation to regulations:** Such information is a necessary part of the knowledge base for effort regulation. Different types of regulations with input restrictions result in different reactions from the industry. The restrictions in effort and days available will give opportunities to maximise revenues without increases in costs. Under TAC regulation there is an incentive to increase costs and investments to increase the competitive power of individual vessels in order to obtain as big a share as possible of the common catch quota [3]. Effort regulation creates incentives to develop production factors to maximise catch per day by, for example, upgrading fishing power—investments in more efficient vessels increase capacity (capital stuffing) and lead to increase in costs (for example, the Faeroese experience detailed
in Section 7.3). TAE, accordingly, has to be corrected for increased fishing power. This means the requirement to have an input regulation system that stabilises fishing pressure. Thus, either models have to be established which can provide the knowledge base for this for the advisory system to management, or direct information about efficiency development must be obtained. Limitations in capacity may have the opposite effect, with low investments in vessel efficiency. Consequently, the economic effect can vary between different types of input regulation, which is also dependent on how resource availability affects incentives to invest or decommission. Accordingly, the knowledge base also has to comprise expected reaction patterns of fishermen to different types of regulations [4].

It is through the selected examples shown that necessary knowledge base of discards, over-capacity, and compliance must be available in an input regulation system [5]. Discards are linked to regulations. Effort regulation is introduced with an explicit purpose of reducing discards (for example, in the Faeroese and Australian cases in Sections 7.3 and 7.4). Discards, which a TAC system encourages, are considered to be wasteful by the fishermen. Consequently, the legitimacy and compliance of regulations concerning discards are often low. The fishermen have an incentive to utilise the quota as efficiently as possible, by discarding less valuable size groups and species, and those fish caught in excess of the TAC. In contrast, an effort regulation system will reduce discard as the incentives to discard will be reduced. This will, furthermore improve the quality of catch data and, thus, be of benefit to the knowledge base of management. However, discards resulting from economic high-grading may continue to be a problem in an effort regulation system, as the fishermen will choose to optimise the value of their catch. Yet, this is often prevented—as in the Australian example (Section 7.4)—as fishing costs are too high relative to the differential between the price of different catch categories. Over capacity under an effort regulation will probably not be as big a problem with respect to discards as the over-capacity will not be activated here. With respect to compliance under effort regulation, it is considered that control under this system is simpler because it is easier to observe the activity of vessels than to monitor catch. One option is to base knowledge of fleet activities on information from a vessel monitoring system (VMS) (for example, in the Faeroese and Australian cases in Sections 7.3 and 7.4).

Global examples of effort regulation and their knowledge base: Different types of capacity and effort-based management systems are described using selected global examples. These are characterised by the type and content of the knowledge base available, used and needed in the respective fishery systems, as well as by the evaluated gains obtained by input management compared to alternative systems.

1) The demersal gadoid fisheries of the Faeroe Islands: This fishery is managed through a uni-jurisdictional effort and capacity regulation system. The Faeroese effort regulation system uses individual and transferable effort quotas based on a fleet segmentation and fleet-based access rights to designated fishing areas. This system has the data available for performing virtual population analysis (VPA) based biological stock assessments for targeted species, but only limited catch and effort information is presently available. In this system, catches are converted to a fleet-based TAE by evaluating the historical relationship between effort and partial fishing mortality, but changes in the development of the fishing efficiency are not taken into account.
Managers have access to some fleet-based economic information that may be used in an evaluation of the fishery. However, the compilation is only partly developed. Thus, the system has a high level of stock and fishery information and knowledge base. The choice of capacity and effort regulation in this system is mainly due to the perceived advantages relating to compliance with regulations and control aspects of regulations. The system is also widely accepted because of the involvement of the industry in the advisory process.

(2) The Australian Northern Prawn mixed fisheries: These fisheries are regulated through restrictions on overall vessel capacity, which have licensed access to the fishery through transferable statutory fishing rights, and without maximum activity regulations for licensed vessels. The system uses individual and transferable vessel-based capacity units, defined partly from vessel and engine size, as well as fishing gear size. The extensive catch and effort data available is used in surplus production models and stock-recruitment models for stock assessment of target species, while there is also here only limited availability of information needed for performing VPA-based assessment. Bio-economic assessment of the fisheries is performed using fleet-based economic information. Consequently, there is also here a relatively high level of stock and fishery information and knowledge base. The system is characterised by direct industry and stakeholder involvement in the advisory and management processes and bodies (co-management). This uni-jurisdictional system has always been input regulated. This is mainly because of evaluated advantages of using effort regulation in relation to discard behaviour, the quality of catch and effort data, and limited biological data to assess the large amounts and many by-catch species caught in the prawn trawls, as well as for historical reasons.

(3) The mussel fisheries in Denmark: The Danish mussel fishery is single species and single fleet-based, and is carried out in three areas of which one is by far the most important. These fisheries are primarily regulated in a uni-jurisdictional system through restrictions on access and overall capacity. The system involves co-management where quotas are fixed by the national authorities after consultation with the industry (fishermen). Stock estimates are performed; however, the system is characterised by an absence of VPA-based biological stock assessment, recruitment and forecast models—limited biological knowledge. There is a high knowledge level regarding economic aspects of the fisheries, which are very profitable, and the system has extensive catch and effort and fishery information available, which are used to a certain degree. A capacity regulation system is employed with access to the fishery being restricted using a license system. There is no overall TAC. Daily and weekly vessel quota restrictions are imposed to enhance more equally distributed landings. These quotas are not based on a general TAC, but rather derived from the aim to distribute effort equally. There is little over-capacity in the mussel fisheries fleet, which is reflected in very high economic returns to the invested capital.

(4) The Mediterranean hake mixed fisheries are regulated in a multi-jurisdictional management system comprising EU and non-EU countries. This system can be characterised as a typical relatively data poor example, mainly in terms of fisheries information, including biological data. Hence, for most management units (The General Fisheries Commission for the Mediterranean has defined 28 management
units in the area) there are not sufficient data to perform appropriate VPA-based biological stock assessments, and any results are very uncertain. Only limited catch and effort information is available, but there is some information on overall fleet capacities and, thus, on the maximum possible performable effort. Also fleet-based economic information is diverse between countries. Based on this situation, these fishery systems are capacity regulated in relation to overall entry and exit of fishery (aiming not to increase tonnage), followed by certain technical measures (minimum landing sizes, spatiotemporal fishing closures). Consequently, this system is regulated through adaptive management to the data situation.

(5) The North Sea demersal mixed human-consumption fisheries under the existing EU fishery system are characterised by a multi-jurisdictional TAC and catch quota management system, with a further TAE regulation on top on that. The fishery system for target stocks and many by-catch demersal stocks in the North Sea has data and models available for performing VPA-based biological stock assessments. Also, it has extensive catch and effort information available, as well as fleet-based economic information. Thus, this system also possesses a high level of stock and fishery information and knowledge base. The choice of having an effort regulation system on top of the traditional catch quota system here is mainly a political choice in relation to the need for further restrictions on the fleet, especially in relation to stock recovery plans. This example focuses on the knowledge base and problems of allocating harvest control rules to mixed fisheries in the advisory and management processes for these fisheries.

7.3 THE EXAMPLE OF THE DEMERSAL GADOID FISHERIES IN THE FAEROE ISLANDS

The Faeroese society experienced a deep economic crisis in the late 1980s and the early 1990s that caused a significant deficit in the Home Rule Government’s budget and created difficulties for the main commercial banks to meet solvency standards. As the economy is narrowly dependent on fisheries and aquaculture, the crisis deepened due to a significant set back in the fisheries as the main demersal fish stocks were at historically low levels in the early 1990s. Consequently, they recognised the need to develop a more efficient regulatory system for fisheries, which had, until then, been heavily subsidised and regulated only by technical measures (closed areas and mesh size regulations).

The Faeroese Home Rule Government, in agreement with the Danish Government, established a commission with the task of exploring regulatory approaches that would allow for an optimal and sustainable resource utilisation enabling the fleet economy to be sustained on regular market terms. The commission report [6] recommended a TAC-based ITQ regulation system covering all vessels above 20 BT (total tonnage), while artisanal fisheries were to be regulated through a separate catch ceiling.

The resulting regulation, introduced in 1995, was met with serious resistance from industry, and from parts of the political system, on the grounds that it resulted in significant discarding and landings of ‘black fish’. As a response to this resistance to the ITQ system, a new commission was established with the purpose of evaluating alternative regulatory
measures that could mitigate the problems. The report of the second commission [7] recommended the introduction of a fleet-based effort regulation, supplemented by a restrictive system of fleet-based area closures. The recommendations were codified in a new fishing law introduced for the fishing year 1996–97 [8]. At that time the Faeroese fleet was reduced significantly due to the economic crises.

7.3.1 Management objectives

The objectives of the Faeroese fisheries regulations are only sparsely sketched in official legal documents. The fishing law [8] notes specifically that management shall take place by use of capacity regulation and that the capacity should be frozen at the 1995 level—after the significant fleet reduction resulting from the economic crises of the preceding years.

Supplementary information on the objectives deemed important at the time of the introduction of the effort regulation may be deduced from the Terms of Reference (ToR) of the second (1995) Home Rule commission [7]. This document stresses the importance of including both economic and biological objectives (“evaluate regulatory measures within the political objectives on biological and economic sustainability”) and to consider the pros and cons of various regulation schemes with regards to “biological, economic and controllability perspectives”. The later report [7] presumably reflects the discussion of the day, which contrasted the recommended effort regulation with the unacceptable ITQ regime, where it is noted that:

- An effort regulation removes incentives for discarding and black landings
- That this may lead to more reliable catch information for stock assessment
- That an effort regulation is more robust than a TAC regulation being
- less dependent on the recent and most uncertain stock estimates
- That an effort regulation would gain a wider acceptance from industry

Several weaknesses of an effort regime were, however, identified. For example, the problems relating to efficiency increases due to technological development and the lack of tools to direct fishing between species occurring together.

7.3.2 The regulation framework

The Faeroese effort regulations are implemented through the use of individual and transferable effort quotas. The effort regulation applies only to the fisheries off the Faeroe Islands to those vessels that are permitted to fish on continental shelf areas within the Faeroese EEZ. The vessels fishing inside the EEZ, and subject to effort regulation, mainly target cod, haddock and saithe.

Corner stones in the regulations are:

1. A rigid fleet segmentation based on vessel size and fishing gear that is used to distribute and control effort and to regulate fishing access between areas. The fleet segmentation includes: (1) large pair-trawlers; (2) large long-liners; (3) medium size vessels (15-110 BT) subdivided by size and gears; and (4) artisanal vessels that only
include hook fishing (jigs and long line), subdivided into full-time and part-time fishermen.

(2) A capacity regulation scheme aims to freeze the capacity within each fleet segment to its 1997 level. Capacity is regulated through a licensing system. Rules that allow for renewal of vessels (restricting vessel size changes) and for merging (purchasing) of capacity are detailed by governmental orders.

(3) An effort regulation where the effort is regulated through annual laws agreed in Parliament. The law provides the total effort (measured by sea days) for each of the fleet segments. For the non-artisanal fleet segments, the total effort is subsequently allocated evenly between the vessels in each of the fleet segments. For the artisanal fleet, 60% of the total effort is allocated to the full-time fishermen who receive individual and equal-sized effort quotas. The part-time fishermen, in contrast, fish on a common effort quota, implying that this fishery is halted when that quota is used.

(4) Tradability of the effort. The sea days are tradable within and between the non-artisanal fleet segments. For the between-segment trade there is a need to consider differences in fishing power across vessel sizes and gear types. The fishing power differences are accounted for through effort conversion keys detailed in governmental orders. These, for example, stipulate that one large-longliner sea day equals 1.33 trawl sea days for a medium sized trawler.

(5) A regulation of the fishing pattern based on access rights to fishing areas. The regulation is detailed by fleet segments and gives priority to smaller vessels and the hook fisheries. The artisanal fleet (hook fisheries only), thus, has no access limitations, except for some smaller areas that are closed to all fisheries during spawning time. The medium-sized liners are excluded from areas within 6 nm of the coast and the largest liners from the areas within the 12 nm line. Trawlers are, in general, excluded from the entire continental shelf area through a number of boxes closed to trawling. There is, however, a derogation available for the smaller trawlers (approximately ten vessels) that allows a summer fishery targeting flatfish, and using trawls with sorting grids within a number of designated shelf areas. Gill netting is restricted to deep-water fisheries targeting Greenland halibut and monkfish.

(6) Additional technical regulation measures includes:

- Mesh size regulations that are usually detailed by fisheries and associated with minimum catches of the targeted species groups;
- A general discard ban, including real-time rules for changes of fishing area when discards occur;
- Minimum landing sizes.

7.3.3 The institutional setup of knowledge production

7.3.3.1 Data sources

The data used for stock assessments (total catch and commercial fishery effort) is compiled by the Ministry of Fisheries and Maritime Affairs and by FRI (the Faeroese Fisheries Laboratory), sampling for age distribution of catches, maturity and mean weights. Routine
resource abundance surveys (fishery independent abundance indicators) covering the major demersal stocks are conducted twice annually. For various practical reasons, the ministries catch/effort statistics are not computerised for assessments purposes and commercial stock abundance indicators are, therefore, restricted to selected vessels within the most important fleet segments where the catch/effort data is separately compiled by FRI.

In spite of the explicit objective of economic sustainability stipulated in the preparatory works of the present Fisheries legislation [7], there is no public database available on the economic performance of the fleets. Some economic data on the largest vessels (roughly above 100 GT), have been compiled by a private audit company in a recent evaluation of the Faeroese management system [9].

7.3.3.2 From data to advice in the production of knowledge: approaches to analysis and advice production

Evaluations of stock sizes (spawning stock biomass) and their exploitation (fishing mortalities) are carried out by single stock analytical assessment approaches—mainly the XSA-model [10]—that are calibrated with resource abundance survey time series and/or commercial fleet catch rate time series.

The Faeroese Home Rule government asks the International Council for Exploration of the Sea (ICES) to provide advice on exploitation rates. This biological advice from ICES is established and provided through international assessment working groups and ICES’ Advisory Committee for Fishery Management (ACFM). ICES have noted that detailed catch rate information at the fleet segment level are needed if ICES is to be able to thoroughly evaluate the Faeroese effort regulation.

ICES relate their advice to established biological reference points for target and limit biomasses and fishing mortalities on stock basis, according to their interpretation of the Precautionary Approach [11]. It is alternatively possible to relate stock sizes and exploitation rates to Faeroese Harvest Control Laws (HCLs), used to evaluate medium-term simulations by the second Governmental commission [7]. These HCLs are, however, not codified in the legislation.

7.3.3.3 The decision-making process and communication of advice

The decision-making process focuses on the number of sea days available for each fleet segment. The Faeroese fishing law stipulates that this number should be based on simultaneous advice from the Fisheries Laboratory and from the Sea Days Committee of the Faeroe Islands (Fig. 7.1). The research institute bases its advice on ICES’ advice, as described above; whereas, the fishing industry’s advice is developed within the Sea Days Committee, that consists of owners and employees representing the various fishing fleets, with a chairman appointed by the government. On this basis, the Ministry of Fisheries of the Home Rule Government of the Faeroe Islands make a proposal on the number of sea days that is later amended and passed as a law.

7.3.3.4 Implementation: control and enforcement.

The regulation requires that the Faeroese (national) fishery inspection force continuously monitor the use of sea days by each vessel. Vessels are, therefore, requested by law to
report when they leave or return to harbours. Until 2003, this was done by telephone but is now based on VMS information. Similarly, selling and buying of sea days must be approved by fishery inspectors who converts sea days across different vessel sizes and types (fleets). Vessel owners are continuously informed of their remaining “balance” of sea days by the authorities.

### 7.3.3.5 Compliance and acceptance

There are no incentives for non-reporting of catches and only limited incentives for discarding catch. High-grading discards has been evaluated to be insignificant in the Faeroese fisheries under this effort management system. There is, however, an incentive for under-reporting effort used. It is generally believed that a significant under-reporting of effort occurred before the introduction of the VMS control system.

The effort regulation system has attained a wide acceptance by industry. This is partly due to the fact that the industry had its say when the system was introduced and that the views of industry are habitually heard in relation to the setting of the annual sea days (effort quotas) in the advisory process. Significantly, no reduction in the number of sea days has taken place since the system was introduced in 1996. Consequently, the high acceptance is presumably also influenced by the fact that both the biological and the market systems have been very favourable for most of the period since the introduction of the effort regulation system—featuring both good recruitment in the most important demersal fish species (cod, haddock and saithe) and a doubling of cod and haddock prices from 1996 to 2000.
The efficiency of the system, and its implementation with respect to both preventing over fishing and enhancing fleet earnings, have not been thoroughly assessed. It may be observed that, although some reductions in the annually allocated sea days have taken place (approximately 8% since 1997), most fleet segments are recorded as having not used their effort allowance. The recent significant decline in cod stock size that led ICES to advise a significant reduction in effort is not reflected in a reduction of sea days.

7.4 THE EXAMPLE OF THE AUSTRALIAN NORTHERN PRAWN MIXED FISHERIES

The Northern Prawn Fishery (NPF) is the largest trawl fishery, as well as one of the most valuable and mature fisheries, in Australia. It was first established commercially in 1960. The annual average prawn harvest is 8500 tonnes, with a production value of between AS100–170 million (63–107 million Euro), and estimated annual net return of the fishery between AS19–40 million (12–25 million Euro). The fishery is located off Australia’s northern coast, covering 770,000 square kilometres of ocean. Two main types of prawn are caught in the NPF: tiger and banana prawns. More than 90 percent of the catch is frozen on board trawlers and exported to South-East Asia. (Australian Fisheries Management Authority, [12, 13]).

While the fishery expanded rapidly in the 1970s, tiger prawn catch rates fell during the next decade. Commonwealth Scientific and Industrial Research Organisation (CSIRO) research indicated that excessive fishing effort may have been reducing the spawning stock to critically unsustainable levels, and brown and grooved tiger prawns have been classified as over-fished for several years [12, 14]. In response to this concern, the NPF has undergone restructuring since the late 1980s, from when the objective was a significant reduction in overall fishing effort and capacity within an input management system with transferable statutory fishing property rights and gear unit management. Additionally, indirect input regulations with geographical and temporal fishing closures and gear restrictions have been implemented to protect spawning female tiger prawns. Since this management strategy was adopted, tiger prawns catches have stopped declining but have not recovered [12].

The NPF is a multi-species, multi-fleet fishery and large amounts of by-catch are caught in prawn trawls [15, 16]. Recently, priority has been given to implementing by-catch and by-product catch reduction strategies and regulations, as well as to assess by-catch [12, 17].

7.4.1 Management objectives

Universal management objectives and strategies of ecological sustainability are applied to the NPF in line with precautionary principles. No specific ecologically based management objectives and limit reference points or limit indicators have been set for the NPF or the stocks exploited therein. However, specific objectives for the minimisation of fishing impacts and protection of non-target species (including endangered species such as turtles...
and sea snakes) have been formulated through, for example, the Environment Protection Act (1974) and the Environment Protection and Biodiversity Conservation (EPBC) Act (1999), as well as through the Commonwealth’s Guidelines (legislation requirements) for the Ecologically Sustainable Management of Fisheries (by December 2003) on target, by-product and by-catch species and the broader marine environment. The resulting management strategies are given in a By-Catch Action Plan.

Performance criteria—the measures of success of management strategies in achieving the specific management objectives of the NPF—have, so far, only been evaluated on a single stock basis for the target species, making use of MSY as specific biologically based reference points to ensure a sustainable harvest of stocks. Since 1991, objectives have also included economic efficiency, although no specific socially based objectives and management reference points have been formulated and brought into use for the NPF.

In the tiger prawn fishery, the specific management objective is the target of MSY, which is broken down into the levels of spawning stock \( S_{\text{MSY}} \) and fishing effort \( E_{\text{MSY}} \), which produces maximum yields. Until 2001, the target reference point was \( S_{\text{MSY}} \), while \( E_{\text{MSY}} \) has recently been set as a limit reference point below which serious remedial action has to take place. From 2001, a new more conservative biological target reference point, giving at least a 70% chance that the spawning population will be above \( S_{\text{MSY}} \) after 2006 [12]. Brown and grooved tiger prawns have been classified as overfished for several years [14]. During 2002, AFMA introduced new measures to reduce effort on brown tiger prawns by 40% and on grooved tiger prawns by 25%, achieved by shortening the fishing season as well as by a 25% reduction in the value of the Statutory Fishing Rights (see below) [12, 18, 19].

The management objective for banana prawns is also MSY. Yet, for endeavour prawns a biological reference point, \( F_{\text{REP}} \), is defined in place of MSY because of the linear stock–recruitment relationships, where \( F_{\text{REP}} \) is the fishing mortality associated with recruitments levels that would replace the parent stock [12].

It is unknown whether the apparent failure of the tiger prawn stocks to recover is related to limited management objectives and options, to the serial depletion of stocks, or to the use of MSY, \( B_{\text{MSY}} \) and \( E_{\text{MSY}} \) management targets for stocks whose dynamics are dominated by yearly recruitment variation and where MSY may well give false expectation of stability. Management targets that relate to present conditions rather than to equilibrium conditions (for example, target fishing mortality rate) may better serve intrinsically variable fisheries, such as prawns [12]. Also, an actual technical efficiency increase of the NPF fleet in certain periods may explain the failure to meet management objectives. Efficiency has, over a period, increased in terms of vessel size and engine capacity (being management control units previously, see below). Regulations reducing this type of capacity led, however, to rapid substitution from regulated to unregulated input, such as gear headrope length, by fishermen. This has resulted in the outcome that effective effort has probably not been cut significantly. However, in the most recent period restrictions on gear size, coincident with increasing restrictions on vessel size and engine capacity, has resulted in efficiency decrease. The recent technical efficiency decline indicates that the management objective of increasing economic efficiency has not been realised. [20].
7.4.2 Management instruments

The NPF is a national (Australian) and regional-county managed fishery, extending from the low water mark to the outer edge of the 200nm Australian fishing zone. Under an Offshore Constitutional Settlement (OCS) agreement between the Commonwealth, Western Australia, Northern Territory and Queensland (1988), the NPF, up to the low water mark, is the responsibility of the Commonwealth through AFMA.

Their being community property, the resources of the NPF are managed by limiting entry to the fishery (starting from 1977 [20]) through a series of direct and indirect input controls covering vessel capacity restrictions, gear restrictions, by-catch restrictions, and a system of seasonal, spatial and temporal closures. The Fisheries Administration Act (1991) and the Fisheries Management Act (1991) created the statutory authority model and instrument for fisheries management. The NPF Management Plan 1995 sets out the management objectives, measures, and performance criteria of the fishery, and makes provision for the fishery to be regulated by AFMA directions and for the implementation of the Bycatch Action Plan.

Management has aimed to reduce the overall effective fishing effort significantly by capacity reduction regulations. In 1977, 292 vessel licences were issued, whereas, in 2002, the commercial fleet was reduced to approximately 96 (102 are trawlers allowed to operate in NPF under the present management scheme) [12][20]).

Provision for private secure and transferable access property rights (Statutory Fishing Rights, SFRs) for major commercial fisheries, as well as implementation of binding conditions on fishing permits, are key in pursuing efficient harvesting regimes. These are aimed at providing market-based incentives for commercial fishers to enhance and conserve resources and a level of certainty, combined with the flexibility needed in the dynamic environment of fisheries management. However, trading in the transferable licences resulted initially in allocation to newer and larger vessels, and, after the licenses were issued, more fishers took up their fishing entitlements [20]. The input regulation of NPF is today based on capacity restrictions with SFRs giving limits on the number of trawlers and the amount of gear permitted. A system was implemented where trawlers were originally ‘unitised’ with respect to fishing power, defined in terms of under-deck hull volume and power of the engine [12]. This system is described here.

A Class B unit which authorised the operation of a boat in the fishery, and a Class A unit, defined as a sum of the kilowatt of engine power and the under-deck hull volume in cubic meters (for example, a boat with a hull volume of 75 cubic metres and engine power of 300 kW would require one Class B unit and 375 Class A units). Small trawlers that, in 1984, did not qualify for the Government ship-building subsidy were issued a minimum of 375 Class A units.

The boat replacement policy required vessel owners to acquire A-units from other licence holders. In 1993, the system was modified so owners wishing to upgrade or introduce a new vessel were obliged to surrender A-units and a vessel licence, resulting in the removal of at least one other vessel from the fleet, to re-allocate their A-units among the vessels they owned, or to purchase A-units from other fishers who then exited the fishery [20].
A 1999–2000 amendment to the Management Plan (1995), tabled in Parliament, introduced gear unit-based management, laying restrictions on the length of the headrope and footrope, which can be varied depending on the status of the prawn stocks [12, 21]. Within this system, each trawler with an entitlement to fish is issued with:

A Class B SFR which permits a boat to fish in the NPF, and a Gear SFR which limits the amount of net a vessel can use (a gear SFR represents 7,500 cm operational headrope and 8,625 cm operational footrope).

Units are transferable and are the ‘currency’ and the property right in the fishery. Voluntary ‘buy-back’ schemes of A-units in 1985 and 1991, and, for example, a 25% reduction in SFR value in 2002 have been used to reduce the number of trawler licenses and fleet capacity [12, 13, 18, 19, 22–24]. Kompas [20] found that average technical efficiency in the NPF was directly constrained by input controls (A-units) and that technical efficiency declined due to input substitution (to gear headrope length). This study indicated the negative effects of input controls on technical efficiency, even in the absence of input substitution.

In addition to direct structural tools within effort regulations, indirect effort regulation measures have been implemented and controlled through the Management Plan (1995). They include technical measures such as spatio-temporal closures: for example, (i) permitting only two open fishing seasons, generally falling between April and May and between August and November; (ii) restrictions of daylight trawling between August and December; as well as (iii) permanent closures of shallow water seagrass beds and other sensitive habitats, including nursery areas for juvenile prawns. Seasonal closures coincide with the spawning and recruitment phases of prawns to ensure protection of pre-spawning female tiger prawns, and to ensure individuals are at an acceptable size for harvesting. Additionally, the length of the seasonal closure depends on actual prawn stock status. Prohibition of daylight trawling also reduces the relative capture of gravid prawns. Furthermore, gear restrictions to reduce fishing effort in the form of limits on net sizes have been used [12, 17, 20].

No direct TAE is set in the NPF and no TAC is calculated.

An estimated 30,000 tonnes of bycatch is discarded in the mixed NPF each year [15]. Typically, less than 20% of the catch is prawns, and the remainder—mostly fish and crustaceans—is discarded because it is not feasible to retain large volumes of low value by-catch [12, 15, 16, 25]. AFMA and the fishing industry, through the NPF Management Advisory Committee (NORMAC) have monitored by-catch [17]. Action Plans (for example, the Commonwealth Policy on Fisheries By-catch and the Northern Prawn Fishery Management Plan (1995)), AFMA Directions, fishermen education programmes, and measures in relation to by-catch reduction include large area closures, compulsory use of turtle excluder devices, use of other technical by-catch reduction devices, a strategy for managing incidental seal catch, and a ban on the take and retention of any products from protected and endangered species, for example, sharks, rays, sawfish, and some larger fish. Catch (and minimum landing size) limits of by-product catch and other by-catch in the NPF have been set by AFMA (for example, on squids, slipper lobsters, scallops) using the terms of the Offshore Constitutional Settlement Agreement between Commonwealth and Queensland, the Northern Territory and Western Australia.
7.4.3 The production of knowledge

7.4.3.1 The institutional setup of knowledge production to management, decision-making processes, and communication of advice

AFMA, being a specialist Commonwealth fisheries management agency, is the statutory authority responsible for the day-to-day management of fisheries for the benefit of the community as a whole and is, thus, a body independent from the Minister. As such, the decision-making process is one step removed from direct political processes. Broader fisheries policy and international negotiations are administered by a smaller group within the Department of Agriculture, Fisheries and Forestry—Australia (AFFA). AFMA includes an expertise-based Board, private access rights-based management, and a strong partnership approach and consultancy (and advertisement) process obligation with key stakeholders. AFMA make direct public recommendations to the Federal Minister for Fisheries and the Parliament through the Corporate Plan, the Annual Operational Plan, and the Annual Report (see Figs. 7.2 and 7.3).

Fig. 7.2. The institutional setup of knowledge production within the NPF and the relationship of AFMA with other agencies [12].
Fig. 7.3. Sources of data and data collection for the NPF, and some routine data processing pathways [12, 81].

The Board of Directors makes the high level decisions on fisheries management and is responsible for setting the policy framework and ensuring that adequate resources and expertise are available to meet AFMA’s legislative obligations. The board represents natural resource management, fishing industry, finance, conservation groups and research.

The AFMA Board is assisted by Management Advisory Committees (MACs) for each fishery, including the NPF (NORMAC), whose members are drawn from AFMA management, science, industry/fishers, and environment/conservation groups, as well as observers from province governments. MACs provide a forum for discussion and consultancy of issues relevant to the fishery and its management, facilitate the flow of information between stakeholders and advice, and make direct management recommendations to the Board.

Also, the AFMA Board is assisted and advised by the Fishery Assessment Groups (FAGs), NPFAG for the NPF, with membership drawn from fishery scientists, fishery
economists, industry, AFMA management, conservation groups, non-governmental organisations, recreational fishing, and other interest groups. FAGs synthesise biological, ecosystem and economic information on Commonwealth fisheries and coordinate, evaluate and regularly undertake fishery and stock assessments in each fishery. Although cooperating closely together, FAGs operate independently from MACs.

Finally, the Board is assisted by an Environment Committee (EA), providing advice on strategies for environmental and conservation issues, reviewing key AFMA environment documents. The EA include AFMA management, industry, science, conservation groups, NGOs, and the Commonwealth’s Environment Agency (Environment Australia). [12, 13].

7.4.3.2 Data sources for knowledge to support fisheries management

A data collection programme covering both fishery-dependent and independent information has been established for the NPF, according to Fisheries Management Regulations. The programme is summarised in Figure 7.3. These data support stock assessment of target species, estimation of composition and abundance of by-catch, and evaluation of interaction with endangered, threatened or protected species. Information is maintained for nine prawn species targeted in the NPF, out of more than 50 prawn species caught. Three species account for almost 80% of the total annual catch [12, 13].

The data collection programme is based on fisheries logbooks, providing daily catch and effort data, as well as information on vessels and gear used and location of fishing activities [13]. Logbook data include quantities of the catch of target species, and, in recent years, also catches of by-product and by-catch species (including information on catch and release of endangered, threatened and protected species). The logbooks give information on both nominal and effective effort (where effort standardisation takes into account effort creep), given as the number of fishing days and number of hauls. Logbooks also contain vessel capacity and gear information. NPF trawler operators have voluntarily recorded the size composition of the catch since 1985 (covering 30–40% of the catch) [26].

Catch and effort data from logbooks, sampling of vessel gear details, and information on general fishing power developments have been used to estimate effective fishing capacity and effort units over time. This includes analyses of data on technological advances in vessel and gear design, and improvements in fish-finding and navigation equipment, as well as developments in technology utilisation in the NPF [27, 28]. Some assessment models incorporate fishing power increases and technological creep in an attempt to produce a comprehensive effort effectiveness analysis model for the NPF [12, 29].

Logbook information is entered into the database Australian Fishing Zone Information System (AFZIS). Furthermore, NPF trawler owner and processor records are obtained for landings data and trans-shipment records used to verify logbook data. [12, 30]. Trans-shipment records contain information about amounts of target and by-product species. Identification, quantification and validation of logbook information of by-catch are further strengthened and obtained through on-board research programmes, scientific and technical observer programmes, crewbased on-board monitoring programmes and scientific surveys [12, 26].

CSIRO has recently (2002) established two annual fishery independent surveys in January and September, the primary aim of which is to obtain independent prawn recruitment indices, and to examine spatial contractions and distributions of the prawn resources.
and the fishery, as well as to evaluate changes in NPF fishing power [31]. Data sampling through CSIRO Marine Research programmes also covers some relevant environmental data, such as the finding that, for example, banana prawn catches are believed to be influenced by environmental conditions (cyclones). Data also cover the distribution of: (i) fishing effort, quantification of catch including discards; (ii) splitting of catch categories and species over time and area to identify spatial stock structure and distribution patterns; (iii) indices of recruitment and environmental factors affecting those; (iv) stomach samplings (predation); (v) mapping and sampling of community composition and habitats (bio-diversity) and trawling impacts on those; (vi) biological production; as well as (vii) physical factors (sediment types, hydrography and wind stress [12, 14, 32–37]).

7.4.3.3 From data to advice: approaches to analysis and advice production

Strategic assessment of the NPF is performed as a part of AFMA’s obligations under the EPBC Act (1999). This covers stock assessments and economically based fishery assessments of the cumulative impacts of the fishery on target species. The strategic assessment is reviewed and revised according to comments received from the public and stakeholders before approval of the final assessment by the AFMA Board [12].

The NPFAG assess stock status and dynamics of NPF target species. The group meets three times a year and publish an annual assessment report. The report includes analysis of previous and current stock assessments, implications of assessments for management, assessment of the economic status of prawn stocks, and analysis of environmental and ecological factors affecting prawn stocks. The assessment models are of the type of single species surplus production or biomass dynamic models calculating MSY, which is broken down into levels of spawning stock at MSY ($S_{MSY}$) and fishing effort at MSY ($E_{MSY}$), as well as Stock-Recruitment models. Each target species is subject to individual assessment. For tiger prawns, several models have been developed [33, 38–40]. The Wang and Die [38] assessment model was based on deterministic growth and deterministic seasonal recruitment patterns, not using size-structured data. Main developments in assessment models for the NPF include a stochastic model used in recent tiger prawn assessments, which provides information about uncertainty levels, as well as estimating $S_{MSY}$ and $E_{MSY}$ [33]. This model performs risk analysis to assess the probability that stocks are being fished at sustainable levels in relation to reference points and performance indicators of stock status (to enable risk management decision). Also a Deriso-Schnute weekly delay-difference model has been applied by Dichmont et al. [41] for stock assessment of tiger prawns.

Assessments of banana prawns and tiger prawns include some environmental data on climatic factors where higher than average rainfall is probably linked to higher than average banana prawn catches during the preceding summer and vice versa [42, 43]. This is coupled to La Nina and Cyclones. Cyclones have also destroyed coastal sea grass beds, which are important nursery areas for tiger prawns, having had a dramatic impact on the NPF over time [44]. The link to environmental factors has led to the application of stock prediction style assessments (prediction models) for the banana prawns. Previously, yield-per-recruit-models were used for certain banana prawn stocks [45], and historic
stock sizes and fishing mortality rates have been estimated by Die and Loneragan [46] using historic logbook data. The establishment of a new assessment and evaluation of the influence of the environment and fishing on long-term catches of banana prawns indicates a relationship between spawning stock size and catches, and that environmental factors do not alone determine catches [47]. As yet, there is still no conclusive evidence that effort influences stock abundance in this fishery [20].

Discards of prawns are not included in stock assessments as discarding is not systematic and has not been considered important in recent years because all ships now have freezing facilities on board, and also because the input regulation gives no economic incentive for discarding prawns [12, 48].

The Australian Bureau of Agricultural and Resource Economics (ABARE) undertake bio-economic assessments of the NPF every 3–5 years [12, 49, 50]. The key economic question in this assessment is whether the fishery results in the maximisation of the resource rent—this being the long run excess of income from the fishery after fishing and management costs. As a proxy for the resource rent, the net return of the fishery is calculated. The assessment overall is based on results from two directions: (1) estimation of technical efficiency in the fishery as a measure of how much the potential output is realised given inputs and technology; (2) use of a bio-economic model of the fishery to estimate economic efficiency in the form of impacts on net economic returns (and profit) to the fishery of alternative management arrangements. The economic assessment of the NPF shows a lower net return than could be achieved because too many vessels are expending too much effort to catch rather limited resources [49]. A bioeconomic model was produced in 2001 by ABARE to assess the economic impact of changes in management (for example, variable closures), prices or costs on both the profitability and the NPF fleet structure, as well as to improve understanding of the spatial dimension of fishing activity and catch [51]. The model draws on results from biological models (stock-recruitment models), adding statistically estimated behavioural equations to fishing effort, on which the economic and institutional framework of the fishery was imposed within a stochastic optimisation framework. The model output evaluates whether stock levels support (or not) MSY. Furthermore, a multi-species simulation model of NPF to assess the biological and economic effectiveness of seasonal closures in relation to yield, income, net operating income, and spawning stock indices per recruit, has been developed by Somers and Wang [52].

EA has, in conjunction with AFMA, developed guidelines for Assessing the Ecological Sustainability of Commercial Fisheries. However, no ecologically based assessment is performed for the system(s) including NPF stocks and fisheries, and no consistent assessments of by-product species and the effect of NPF on stock dynamics for those species are performed. Stock assessment of by-catch species in the NPF has been proved to be difficult because of the diversity of by-catch, because most species are rarely caught and there is limited knowledge on the biology of many of the species [12]. Research is ongoing to evaluate (i) the impact of discarding; (ii) the composition of NPF by-products and by-catch in relation to testing of different by-catch reduction and exclusion devices, as well as for optimal selection of closed areas; (iii) selection of marine protected areas; (iv) impacts of trawling on benthic species assemblages; (v) impact of cyclones; and (vi) potential management strategies for conserving biodiversity values [12, 16, 17, 25, 48, 53–58].
7.4.3.4 Implementation: control and enforcement. Compliance

AFMA coordinates compliance arrangements for Commonwealth fisheries and develops Compliance Plans for each major fishery. AFMA evaluates both the potential risks of non-compliance with management and the potential impact and consequences of non-compliance. Strategies to manage these risks, including activities, performance measures, and costs of activities, are then incorporated into the plans. Specific compliance functions (implementation of control and enforcement) in the field are undertaken by State and Territory fisheries and police officers employed by AFMA. The compliance plans are translated into Service Level Agreements made with States/the Northern Territory, detailing tasks of the officers. Together the Compliance Plans and the Service Level Agreements make up the Compliance Programme for each fishery.

In relation to enforcement of regulations (for example, spatial and temporal closures) all trawlers are required to carry and operate an integrated computer VMS, this being a mean of ensuring compliance with closures. Enforcement also includes regular random vessel and at-sea-inspections supported by an aerial surveillance programme to inspect gears, vessels, logbooks, trans-shipment records, catch composition with respect to size regulations and by-catch limits, and possession bans. This is additionally supported with in-port-measuring and gear certification.

7.4.3.5 External constraints

In addition to management closures of the NPF, there are a number of other areas protected by legislation located within the NPF area (marine protected areas). Most of these are in areas that are already closed off to any fishing activity, are close to the shore, or are part of terrestrial parks [12].

Industry are required to pay levies for recoverable management of the fishery and for a part of the research costs for the fishery ("users pay policy"), as well as for a part of the ‘buy-back-schemes’ of fishing licenses.

7.5 THE EXAMPLE OF THE MUSSEL FISHERIES IN DENMARK

The mussel fishery in Denmark is a single species, single fleet fishery for blue mussels supplemented by smaller catches of oysters and cockles. By-catches of fish are small. The fishing vessels are relatively small and all use dredges. The fishery takes place close to the shore or in inlets. The fishery is carried out as a one-day fishery.

The Danish mussel fishery is executed in three main areas. The largest area in terms of landings is Limfjorden, followed by the East Coast of Jutland, and then the Wadden Sea in the south east part of the North Sea. The annual landings of mussels in all three areas vary over time but are around 100,000 tonnes including shells, at a value around 14 million EURO (2003). The total number of vessels in the fishery is restricted to 64 (2003), of which two fish in a small inlet and are left out in the following text.

The fisheries are managed by exclusive entry licenses, which include clear specifications of the capacity of vessels. Sea days restrictions are not used. However, catch limits
Effort and capacity-based fisheries management

7.5.1 Management objectives

The Danish management objectives are derived from those of the Common Fisheries Policy (CFP) of the EU. The revised CFP states (Council Regulation (EC) No. 2371/2002 of 20th December 2002):

“The CFP shall ensure exploitation of living aquatic resources that provides sustainable economic, environmental and social conditions”.

“For this purpose, the Community shall apply the precautionary approach in taking measures designed to protect and conserve living aquatic resources, to provide for their sustainable exploitation and to minimise the impact of fishing activities on marine ecosystems. It shall aim at a progressive implementation of an eco-system-based approach to fisheries management. Also, it shall aim to contribute to efficient fishing activities within an economically viable and competitive fisheries and aquaculture industry, providing a fair standard of living for those who depend on fishing activities and taking into account the interests of consumers”.

As this makes clear, European fisheries management objectives are specified in broad terms, taking into account that the exploitation of the resources must take place with a view to the global environment, including the fishery, birds, and use of fishing grounds for leisure purposes and commercial traffic. Tying in with this approach, the objectives of the management of the Danish mussel fishery aim at an environmental, resource, and socio-economic sustainable use of the mussel stocks. There is no specific mentioning of effort or capacity management for this particular fishery. However, in the Danish legal notices issued by the Government, capacity regulation is specified clearly.

7.5.2 Management instruments

The three geographic areas in which the mussel fishery takes place are, in general, managed in the same way, although there are differences in the details. In the following text, emphasis is placed on the management instruments used for Limfjorden, the most important mussel fishery. The main management instrument in force is capacity restrictions, used in combination with individual vessel quotas. The numbers of permits (licenses) are restricted to 62, and there are restrictions on the engine power. For Limfjorden, further restrictions are imposed on the size (the hold capacity) in terms of gross tonnes (GT) of the vessel. In agreement with the fishermen, weekly and daily quotas per vessel are fixed by the Danish Ministry for Food, Agriculture, and Fisheries (effectively by the Danish Fisheries Directorate). The quotas are fixed in gross quota terms—meaning mussels with shell including mud, sand and other items caught. A minimum landing and exploitation size of mussels is set as well.

An outline of the most important management instruments is shown in Table 7.1. The rank of the management instruments is entry restrictions, capacity (in terms of engine per vessel per time period are applied. In the Wadden Sea the fishery is managed in a tri-national agreement, while the fisheries in the other areas are managed nationally.
Table 7.1 Restrictions on the mussel fishery in Denmark.

<table>
<thead>
<tr>
<th>Area</th>
<th>Level of landings incl. shell (tonnes)</th>
<th>Production limits</th>
<th>Effort restrictions</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Week quota per vessel b (tonnes)</td>
<td>Day quota per vessel b (tonnes)</td>
</tr>
<tr>
<td>The Limfjorden</td>
<td>80,000</td>
<td>85</td>
<td>30</td>
</tr>
<tr>
<td>Kattegat/Small Belt</td>
<td>20,000</td>
<td>270</td>
<td>None</td>
</tr>
<tr>
<td>Wadden Sea</td>
<td>5,000</td>
<td>75</td>
<td>40</td>
</tr>
</tbody>
</table>

Source: [82]

\(a\) ICES area 22A.
\(b\) Including mud and sand brought on board.

power, length, breadth, draught, and GT), and individual quotas. Finally, it is not permitted to fish on Sundays.

Although stock estimates are carried out, these estimates serve as indicators of the sustainability of the stocks and not the determination of quotas. Basically, the weekly and daily quotas are determined ‘backwards’ in the sense that the hold capacity of the vessels forms the basis for the possible weekly and daily catches.

There is a strong element of co-management, in terms of direct involvement of the fishermen/industry in the management of the fishery. Formerly, the weekly quota was set at 110 tonnes, but the fishermen argued for a lower rate, which was accepted by the Ministry. The fishery takes place a limited number of weeks every year, and the fishermen themselves agree on the closures. The closures take place in the winter where the meat content of the mussels is low, and in the summer period where the risk of algae blooming and, thus, the risk of food poisoning from the mussels is high. In Limfjorden, the fishery is conducted over less than 30 weeks a year, while in the other areas the fishery is often stopped in most of June, July and August. It should be noted that certain areas within the main fishing grounds are closed to the fishery. The reason for this is that these areas serve as nursery grounds, bird sanctuaries, or shipping routes for commercial vessels and leisure boats. The Ministry of Food, Agriculture, and Fisheries, in cooperation with the Ministry of Environment and the Ministry of Transport, select these closed areas.

Finally, a management instrument not used actively at the moment should be highlighted. This is that economic information has been collected on a sample basis since 1995. This instrument provides information on earnings and costs, which are used to determine the return on labour and capital. These rates of return show the economic performance of the fishery. The mussel fisheries have been managed in the current way since the commencement of the economic information, and the rates of return have been very high in the whole period compared to all other fisheries within the European Union for which information exists. The rate of return is a useful indicator for the viability of the fishery in terms of resource sustainability and fishing capacity used to exploit the resources. The high rate of return on capital indicates that no over-capacity problems persist in the mussel fisheries.
7.5.3 The production of knowledge

7.5.3.1 Institutional setup

The Danish Ministry of Food, Agriculture and Fisheries (the Danish Fisheries Directorate) holds the general responsibility for the management of the mussel fishery, having responsibility for issuing licenses and laying down the legal conditions for the execution of the fishery, including the rules for the vessels. Another directorate, the Danish Veterinarian and Food Administration, is responsible for the monitoring of the quality and the food safety of the mussels and, thus, the time closures, which are quite important because of the risk of food poisoning in very warm periods. The Danish Environmental Protection Agency (under the Danish Ministry of Environment) is responsible for the protection of the marine environment in inlets or close to the shore, which the fishery impacts.

The Danish Institute for Fisheries Research carries out the production of biological information for fisheries research with respect to assessments of the stocks and other types of biological information. The Danish Food and Resource Economics Institute collects cost and earnings information.

In this particular fishery the fishermen’s role as co-managers should be emphasised. Because of the limited entry, the number of agents in the fishery is relatively small in the different areas and, thus, cooperation among fishermen is easy to establish. Further, as the fishery is characterised by an unusually good economic performance and rather equal income distribution, these elements must be regarded, if not a prerequisite for good co-management, at least very helpful for the success of the system.

7.5.3.2 Data sources

A very detailed submission of catch data from the vessels (catch per unit effort) and the processing industry (amounts landed) to the Fisheries Directorate helps the Directorate to manage and monitor the fishery. This information is not, however, used in association with assessment of the biomass. The fishermen are charged for the samples taken by the authorities to monitor the quality of the mussels. Further, the fishermen execute trial fisheries to test the meat content of the mussels. This type of information is used by the fishermen to determine when to start and when to stop the fishery and hence to limit incentives to ‘rush for catches’. Finally, there is a comprehensive relaying of undersized mussels carried out and financed by the fishermen. The relay of mussels in Limfjorden is around one third of the total catches [59].

The Danish Institute for Fisheries Research carries out biomass estimation every year. The assessments are based on trial fisheries in selected areas by a research vessel, from which the size of the biomass is estimated [60, 61]. The procedure is a standard procedure used since 1993. For every dredge haul, the number, weight, and length distribution of the mussels is determined. The stock abundance is calculated, turning the catch of the haul into catches per square metre by use of an equation in which the efficiency of the gear is a function of the density of the stock (swept area method). From this method indication of recruitment to the stock is also determined.

Economic surveys are carried out by the Danish Institute of Food Economics. Based on a sample of around 25 per cent of the 64 vessels (in 2003) costs and earnings are
collected from the auditors of the vessels, and the economic performance is calculated and presented in term of a number of economic indicators.

7.5.3.3 From data to advice

No general quotas are set for the mussel fishery. The capacity restrictions are sufficient to restrict effort to a level ensuring good economic viability and a biomass above critical biological limits. There are no restrictions on sea days, and the weekly and the daily quotas per vessel are calculated from the hold capacity of the vessels as a supplementary measure. The hold capacity is in the range of 15–30 tonnes, and the daily and weekly quotas are agreed with the fishermen with the aim to level out landings. The fishermen themselves determine the number of fishing days, which in the season is determined by the daily and weekly quotas, and the fishermen decide when the season will start and when it ends. Therefore, the fishery is, in its essence, an effort-restricted fishery. The total catch, at around 100,000 tonnes a year, is much lower than the estimated annual stock production, estimated at 0.5 times biomass [62], even though the biomass assessments show a decrease in abundance over the last ten years. The decrease in biomass abundance could be explained by several reasons, but fishing does not seem to be the most important reason. Deoxygenation, as an environmental influencing factor, is regarded as the most important reason [63], and that seems to be an increasing phenomenon and problem. Deoxygenation is, to some extent, outside the control of the authorities in the short and medium term. Therefore, it could be argued that the current catches and fishing mortality, which are within the control of the authorities, are too high in combination with the natural mortality caused by natural predation from birds and by deoxygenation. Despite this, the stock size, although predicted to decrease, is assessed to be above safe biological limits of biomass. The biological limit reference point based on stock biomass is an evaluated limit and not based on statistical estimations.

Fishing power changes over time in the mussel fleet has not been investigated, and no analyses of fishing power have been performed for this fleet from which results could be included in the advice and management of capacity and activity.

7.5.3.4 The decision-making process

The Danish mussel fishery has, for more than ten years, been subject to strict capacity restriction. This is combined with responsible co-management—through participation of the fishermen in the decision-making process—in terms of activity regulation on when to perform the fishery and how the daily and weekly quotas are calculated, supported by a very high return on capital, above 40% [64, 65], no further actions have been taken to manage the fishery. Although the quotas are binding on a weekly basis (not often on a daily basis), the fishery is executed for only 30 weeks (in Limfjorden) and around 40 weeks in the other areas. This means that there is space for even higher landings, which, however, have not occurred.

7.5.3.5 Control and enforcement and compliance

The Danish mussel fishery falls within the remit of the EU enforcement policy, and is, therefore, subjected to landings control conducted by inspectors who visit landing places
and monitor the fishery at sea. Compliance with regulations among fishers is high, which very likely is enhanced by the strongly restricted entrance and the relatively small number of vessels that execute this fishery, as well as by the high profits. It should be noted that annual landings could be higher if the mussels were exploited in the months where the fishery, in agreement between the fishermen and the Ministry, is closed. This is not the case now and this is evidence of good co-management and compliance. The transparency of the fishery is good. The vessels in the fleet are relatively homogeneous, and the fishery takes place close to the shore. The economic performance of the entire mussel fleet is extremely good, which is caused by proper management. All these characteristics of the fishery provide incentives to the fishermen for good co-management.

7.6 THE EXAMPLE OF THE MEDITERRANEAN HAKE MIXED FISHERIES

Hake is one of the main target species of several multi-species Mediterranean fisheries and especially of the demersal trawlers exploiting the resources on the Mediterranean continental slope. The General Fisheries Commission for the Mediterranean (GFCM) has defined 28 management units for the demersal species of the Mediterranean basin and hake is exploited in virtually all units.

7.6.1 Management objectives

Although, in general, the overall management objectives for the EU as specified above apply, the mixed fisheries structure, featuring multi-species and multi-gear demersal fisheries, in the Mediterranean presents difficulties when it comes to applying specific management measures for a given species [66]. These characteristics of the fishery complicate management of hake stocks, as any management approach to be implemented will not only affect hake, but also other target species. This is further complicated by the lack of sufficient data (a typical “data poor” situation) to produce stock and fisheries assessment results capable of supporting actual management decisions. Consequently, there are no explicit management objectives for the hake stocks and the mixed hake fisheries in the Mediterranean for any of the management units. Management actions mainly aim towards universal objectives, such as sustainability of resources and protection of by-catch species. One overarching management objective of the Mediterranean’s demersal fisheries is, however, not to increase the existing capacity of the overall Mediterranean fishing fleet.

7.6.2 Management instruments

As is the case for all Mediterranean stocks (apart from bluefin tuna, where catch quotas are applied), management is exclusively based on input regulation, mainly in the form of capacity restrictions managed through a licence system for access rights to the resources, otherwise known as fisheries permissions.
A series of general international and national regulations oriented towards the conservation of all demersal stocks are in place. However, international regulations only apply to EU waters. They include:

1. Control capacity regimes: limitation on national fleets’ horsepower and gross tonnage;
2. Prohibition of bottom trawling within three miles of the coast or in depths less than 50 m (whatever comes first);
3. A minimum of 40 mm for the mesh size of the cod-end of bottom trawls;
4. A minimum landing size of 20 cm.

Regulations at the national level are established by different European and non-European countries and are aimed at the conservation of all demersal stocks, including hake. They include the temporal closures of certain fisheries in order to protect sensitive biological processes, such as spawning and recruitment, and control license systems to limit entry to the demersal fisheries. For example, Greece closed their waters to bottom trawling from June to September, and there are general prohibitions of fishing in place in certain Italian areas during specific time periods (for biological rest).

The first three regulations do not specifically target hake, but they have been adopted for the overall protection of the demersal stocks in the Mediterranean. The first concerns all marine stocks; the second regulation concerns protection of nursery areas; while the third and fourth are aimed at the conservation of juveniles (immature fish).

### 7.6.3 The institutional setup of knowledge production

Management advice is provided through the annual reports of the Scientific Advisory Committee (SAC) of the GFCM and it is based on biological stock assessments for the various units. These assessments are presented independently (there are no formal assessment groups) during the demersal working group meetings of the Sub-Committee on Stock Assessment (SCSA). Based on those assessments, SCSA formulates advice that is revised and adopted in the SAC sessions.

### 7.6.3.1 Data sources for production of knowledge to support fisheries management

Ideally, the application and evaluation of such a management regime demands knowledge on: (a) species biology; (b) the state of the stocks; and (c) the exploitation pattern and economics of the different fleet segments involved in the fisheries. In order to obtain such knowledge, biological, fisheries and economic data are needed. Among others, those data include:

- Size at age;
- Size at first maturity and maturity stage by season;
- Depth distribution by size;
- Catch-effort data and size distribution of the catches by fleet segment and time scale;
- Selectivity by gear type;
- Capacity and operational costs by fleet segment.
The case of Mediterranean hake is a typical “data poor” example, particularly concerning fisheries data. Catch and effort data series are, in most cases, only short-term and they do not cover all fisheries and fleets (they mainly focus on data from trawl fisheries). The data availability situation is even poorer when moving from the western to eastern Mediterranean areas and from the European to non-European countries. Knowledge on the segmentation and exploitation pattern of small-scale fisheries is very limited and the economics of the fleets are also poorly known. And, while, in general, the biological knowledge base is at a higher available information level, there are a lot of questionable aspects to these data and a lack of precise knowledge, regarding such issues as the duration of the reproduction period and the species growth pattern.

Due to the different national data collection schemes that have been applied, the existing historical data are not always comparable between countries. However, since 2002, all EU Member countries have to follow harmonised data collection schemes, following the directives mentioned in EU Regulation 1639/2001. According to the regulation, fisheries data will be gathered by pre-defined gear type and fleet segment. Thus, it is expected that comparable and standardised data will be obtained in the near future, to some degree. However, it seems that the problem of a lack of harmonisation will persist, as there is no comparable regulation being applied in the non-European countries. Several databases exist with mixed hake fisheries data from the different countries, but, again, there is no any global database to provide homogeneous data sets.

7.6.3.2 *From data to advice in the production of knowledge: approaches to analysis and advice production*

Until the beginning of 1980s, surplus production models were applied to assess the state of a few hake stocks in the western Mediterranean, for which historical series of catch and effort data exist [67]. Later on, yield-per-recruit analysis was applied, and, today, stock assessments are mostly based on VPA and on length-based analysis applied to pseudo-cohorts [68–70]. The data used for stock assessment include the age and/or size distributions of the relevant catches. As there are not any formal assessment groups, the assessments are made independently and are, thus, un-coordinated. Accordingly, the choice of the appropriate methodological approach in the stock assessment is up to the individual scientist who carries out the specific assessment. Such assessments are carried out using different assessment software packages. The FAO-FiSAT [71] and the VIT [72] are the most commonly used software packages for assessment of the Mediterranean demersal stocks.

7.6.3.3 *The decision making process and communication of advice*

Due to insufficient data, stock assessment results are, in general, imprecise and uncertain, and, for several management units, they do not even exist or are outdated. As a consequence, the existing management measures are not really based on assessment outputs, but rather they have been formulated taking into account generic qualitative information that is used for most Mediterranean demersal resources. Such information concerns the reproduction period, the size at first maturity, the distribution pattern of juveniles and gear selectivity studies. Stock assessments play a secondary role, in the sense that they
confirms the presence of over-exploitation, which is, however, a general truth for most resources of the Mediterranean and world seas. In some respects, this approach takes into account the multi-species nature of the Mediterranean demersal fisheries.

7.6.3.4 Implementation: control and enforcement and compliance

The degree of enforcement of the management regime in relation to the hake mixed fisheries in the Mediterranean varies depending on the actual regulation. Regulations that concern spatio-temporal closures for fisheries, limitations on fishing vessel capacity, and minimum mesh sizes of gears are more easily controlled; hence a certain degree of compliance is achieved for these regulations. The full enforcement of minimum landing size measures is more difficult to control, as it demands extensive monitoring of landings. In addition, such a measure does not always have full acceptance among the fishermen, as compliance to this regulation increases the fishery discards, which is considered to be a waste of resources by fishermen.

7.6.3.5 External constraints

Management of the Mediterranean hake resources is affected by various external constraints. There are political constraints that result from a lack of cooperative management, especially between European and non-European countries. Such constraints, in combination with the limited availability of sufficient scientific information, impose problems in defining rational management units, without taking into account national boundaries. They also do not permit the harmonisation of data collection systems between European and non-European countries. Constraints at the level of scientific collaboration, and particularly the lack of formally based stock assessment groups, result in an absence of standardised assessment approaches and complicate the identification of parameter uncertainties that affect assessment results.

7.7 THE EXAMPLE OF THE NORTH SEA DEMERSAL MIXED FISHERIES UNDER THE EXISTING EU FISHERY SYSTEM

7.7.1 Management objectives

The advice given by ICES and the management measures imposed by the EU all have the objective to prevent over-fishing, following the Precautionary Approach. This perspective has been adopted by ICES in its resource evaluation. Currently, ICES’ advice is given on a single stock basis, for all fleets and areas combined. This advice is suitable for setting a total TAC for a single stock, but does not give any indications of appropriate effort levels, nor does it account for the effect of mixed fisheries.

The rules for fixing catch quotas are the “Harvest Control Rules” (HCR). \( F_{pa} \) is the fishing mortality in relation to the “Precautionary Approach” (Figure 7.4). \( B_{pa} \) is the SSB (spawning stock biomass) corresponding to \( F_{pa} \). \( B_{lim} \) is the limit reference point for the stock not having a high probability of producing large year-classes—or the lowest level
Effort and capacity-based fisheries management

Fig. 7.4. The harvest control rule often used in EU management and ICES advice.

of SSB that allows fishing to continue. If SSB gets below $B_{\text{lim}}$, the stock is in immediate
danger of being depleted and fisheries should be stopped or reduced.

ICES advice on target and limit reference points are often used as management objectives
and HCRs by the EU. They apply the HCR on a single species basis. This means
that management and advice do not take into account the fact that, in the North Sea
demersal fisheries, almost all fish are caught in mixed fisheries. “Mixed fishery” means
that a vessel catches several species on the same fishing trip and/or fishing operation. It
is usually impossible to avoid catching certain species together with other species. This
means that a quota on one species has an influence on the catch of all the other species
cought together with the quota-species.

The present ICES advice on resource management is tacitly based on the assumption
that fish stocks do not interact. Interaction between fish stocks can be grouped
into “technical” and “biological” interaction. Technical interaction refers to the fact that
several fleets compete for the same species and one fleet catches several species. Bio-
logical interaction refers to the interaction between stocks created by predation and food
competition.

Most often ICES advice and assessment counts the catch quota against the landings, not
against the actual catches (landings plus discards). For many stocks, the level of discards
is unknown and fishing mortalities are estimated from the landings only. In practice, the
catch quotas set by the EU are, therefore, often not related to actual fishing mortality.

The ultimate goal of EU management and ICES advice is to achieve a fishing mortality
level of $F_{\text{PA}}$ for all stocks. Fishing mortality is created by fishing effort. It is believed that
there is some relationship between effort and fishing mortality. The simplest model is
that of proportionality between fishing mortality and fishing effort. Should that model be
accepted, then the ultimate goal would be to fix the fishing effort of all fleets so that
$F = F_{\text{PA}}$. However, usually, there will be no solution in terms of “Effort”. Any effort
level chosen will almost never correspond to $F_{\text{PA}}$ for all targeted species and stocks in a
mixed fishery.

The simple case of one fleet gives the solution, Effort $= F_{PA}/\text{Catchability}$. That solution
will give the same effort for species A and B only if $F_{\text{PA}}(\text{Species A})/\text{Catchability(\text{Species A})} = F_{\text{PA}}(\text{Species B})/\text{Catchability(\text{Species B})}$. That will usually not be the case for any
combination of species and stocks.
Accordingly, HCR’s must specify the trade-off of reaching $F_{PA}$ for only one species or to compromise between $F_{PA}$’s for different species in the mixed species fishery. One simple approach would be to apply the same factor to all efforts (of all fleets), so that the inequality is met and equality is achieved for only one species. In the following section we introduce another approach.

Restrictions on the number of vessels, by vessel size and type categories, create a natural upper limit to the maximum effort that can be exerted. The Multi-Annual Guidance Programmes (MAGPs) of the EU aimed at bringing fishing effort more into line with available resources. Here, fishing effort is defined as vessel capacity, in both tonnage and engine power, multiplied by activity (days spent at sea). The rationale behind MAGPs is that the available resources should determine the size of the fleet and not, as has often been the case, that the size of TACs be determined by the size of the fleet. The MAGP was implemented in four phases: I (1983–1986); II (1987–1991); III (1992–1996); and IV (1997–2002). A new system for limiting the fishing capacity of the EU fleet was adopted in 2002. It replaced the former MAGPs. The MAGP and its continuation combined with TAC measures have not been sufficient to reduce effort to a sustainable level or to compensate for the technical efficiency increases of fleets (fishing power). Thus, a suite of additional measures has been introduced, notably mesh size regulations, closed areas and limitations on sea-days.

7.7.2 Management instruments

In recent years (starting 2003), the EU has introduced an upper limit on the number of days a vessel can operate per month. This effort limitation scheme has been an integrated part of the recovery plans for certain demersal North Sea stocks, most notably cod [73]. The limitation of sea-days has been combined with a suite of technical management measures and TAC regulations, but the regulations are not independent, as they are all derivatives of one overall principle: the Precautionary Approach.

There are used two types of direct effort management instruments applied for the demersal human consumption fisheries in the North Sea (2005):

- Restriction of the number of vessels (national licence systems).
- Limitation of number of sea-days by gear category.

Several indirect methods of effort limitation are also implemented. These include: (1) Closed areas; (2) Closed seasons; (3) Restriction of gear particulars (for example, mesh size); and (4) Vessel size and engine power restriction. This section deals exclusively with the two direct measures of effort control. TAC restrictions may also have an indirect effect on effort. Exhaustion of catch quotas, however, may result in re-direction of effort, rather than a change in its total value. When evaluating the effect of effort changes it is important to consider the target of the effort in question.

The overall effect of both types of direct effort limitations with respect to reducing fishing mortality works broadly in the same way. They are not necessarily proportional, in the sense that a given relative reduction in the number of fishing vessels results in the same relative reduction in the number of fishing days by all vessels combined. The
difference between the two types of effort management is rather caused by their effect on the reaction of the fishing industry. It is to be expected that limitations in sea-days will lead to investment in more efficient vessels, whereas limitations in capacity may have the opposite effect on investments in vessel efficiency. The economic effect can consequently be different, with limitations on sea-days leading to an increase in the costs of fishing, but not necessarily an increase in resources.

The economic performance of individual vessels will improve when the total number of vessels is reduced, whereas this is not to be expected when sea-days are reduced (at least not in the short-term). The long-term effect of a sea-day reduction is not very obvious, but the expectation is that resources will benefit, and, as a consequence, fisheries will also benefit in the long-term. Reduction in sea-days may or may not lead to better economic performance in the long run, depending on the reaction of the resources to reduced fishing effort. On the other hand, reducing the maximum number of sea-days makes the planning and execution of fishing more difficult and will increase the costs of fishing. Consequently, the profitability of fishing will be reduced, which may have an indirect effect on the capacity: the incentive to invest in new vessels will reduce while the incentive to withdraw will increase.

This assumption of uncertainty of the effects of effort reduction is based on the fact that fluctuations in resources are not only determined by the behaviour of fishing fleets, but on a suite of phenomena which are poorly understood, and perhaps not even recognised by fisheries science.

7.7.3 The institutional setup of knowledge production

The EU Ministers responsible for fisheries assemble every year in December to agree upon management issues for the subsequent year, among which are the fishing quotas to be allocated to each Member State.

The EU Scientific, Technical, and Economic Committee for Fisheries (STECF) assembles at the beginning of November to address management issues and in particular the stock assessments carried out by research institutes associated with ICES. STECF builds its recommendations on reports carried out by two subgroups: the SGRST (Subgroup on Reviews on Stocks) and the SGECA (Subgroup on Economic Assessment). These two working groups convene two weeks before the STECF meeting. The SGRST reviews all relevant information about stock assessment, including recommendations of fishing mortality rates. The information relating to TAC proposals for the coming year—for example, single species recommendations, and ‘mixed fishery’ recommendations provided by the SGRST—are used by SGECA in the economic model EIAA (economic interpretation of ICES ACFM Advice) to calculate the economic repercussions for a number of selected fleet segments. SGRST also provides long-term TAC and spawning stock estimates that are used in the EIAA model to produce a long-term projection of the economic consequences for the fleet segments [74].

Data are collected by national authorities and research institutes and sent to ICES where they are analysed by working groups with respect to stock assessments. ICES’ results are then passed on to the ICES ACFM and the EU STECF and its working groups. These are the primary data used for traditional ICES fish stock assessments, and are supplemented
with effort data. Effort data has also traditionally been used for fish stock assessment, but they are not essential for assessments. However, effort data by vessel category are absolutely necessary for the present analysis and mixed fisheries evaluation.

7.7.3.1 Data sources for knowledge production to support fisheries management

Logbook information on effort and catches, combined with biological samples (age distributions etc.), constitute the inputs for ICES assessment working groups. Effort is measured in the unit of “days” by vessel category (sea-days). The definition of vessel categories (fleets) plays an important role for the analysis. Unlike ICES single-stock assessments, it is essential that catches are given as “mixed catches” by fleet.

7.7.3.2 From data to advice in the production of knowledge: Approaches to analysis and advice production

The STECF (Mixed Fisheries Group) addresses the general problem of mixed fisheries introduced in the previous sections. Appendix 7.A gives a simplified version of a model used by the STECF: the so-called “MTAC” (Mixed Fisheries TAC). Vinther et al. [75], who suggested the use of the MTAC model, give a general introduction to the model. Kraak [76] gives a critical and more complete introduction to the method. STECF [77–79] presents the actual applications of MTAC. Appendix 7.A shows (simplified) how MTAC can be used to calculate capacity and maximum sea-days.

Traditionally, ICES advisers have used the fish stock as the central focus because managers did so in order to maintain the historical international relative stability of the distribution of catch quotas between countries, under the terms of the CFP (EU Common Fisheries Policy). Advice on effort limitations requires the fishing fleets to become the focus. Therefore, switching from stock-based advice to fleet-based advice has been a theoretical challenge for the scientific advisory bodies of ICES and the EU. This section discusses this theory in the light of recent reports of the STECF on “Mixed fisheries” and recent EU legislation.

The MTAC advice seeks to make explicit the terms of the mixed fisheries problem and the nature of the trade-offs necessary to be made by management. It points at the need for a political management decision before scientific advice can proceed in using it for specific advice.

The MTAC approach is to minimise the sum of squares of deviations (SSD) between the target fishing mortalities advised by ICES (and as defined in EU management), and the fishing mortalities adopted by fisheries management. The minimisation of SSD is an approach similar to meeting the inequalities of the foregoing section. It will mean that some of the inequalities will be met and others will be violated. But, overall, it will attempt to minimise the “damage”. That means that we allow some stock to violate the precautionary approach, but we try to minimise those violations.

Meeting this inequality by species may create different priorities for managers. Some species may be regarded as more important than others. Table 7.2 shows the fleets as defined and reported by EU Member nations. The relative importance for each fleet of the landings of each species in 2002 is also indicated. This can be accounted for by the introduction of the so-called “decision weights”. These are species-specific numbers
Table 7.2  Fleets and stocks used as input to the MTAC model [78].

<table>
<thead>
<tr>
<th>Country</th>
<th>Fleet and gears</th>
<th>COD</th>
<th>HAD</th>
<th>NEP</th>
<th>PLE Per mille</th>
<th>POK</th>
<th>SOL</th>
<th>WHG</th>
<th>ALL tons</th>
</tr>
</thead>
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<tr>
<td>BELGIUM</td>
<td>Otter Board Trawl 070–099 and &gt;120</td>
<td>221</td>
<td>37</td>
<td>234</td>
<td>204</td>
<td>77</td>
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<td>1007</td>
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<td>Other</td>
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<td>0</td>
<td>0</td>
<td>39</td>
<td>0</td>
<td>57</td>
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<td>377</td>
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<td></td>
<td>Beam Trawl 080–099</td>
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<td>119</td>
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<td>458</td>
<td>3</td>
<td>173</td>
<td>34</td>
<td>8606</td>
</tr>
<tr>
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<td>126</td>
<td>341</td>
<td>208</td>
<td>89</td>
<td>7</td>
<td>14</td>
<td>8983</td>
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<tr>
<td></td>
<td>Otter Board Trawl 70–99, 100–109, 110–119, &gt;120</td>
<td>146</td>
<td>385</td>
<td>111</td>
<td>222</td>
<td>126</td>
<td>2</td>
<td>7</td>
<td>16797</td>
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<td></td>
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<td>131</td>
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<td>126</td>
<td>9</td>
<td>0</td>
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<tr>
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<td>Other</td>
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<td>569</td>
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<td>26</td>
<td>2</td>
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<td>1</td>
<td>96</td>
<td>80</td>
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<td></td>
<td>Long Lines</td>
<td>761</td>
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<td>0</td>
<td>1</td>
<td>2</td>
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<td>7</td>
<td>101</td>
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<td>0</td>
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<td>1918</td>
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<td></td>
<td>Long Lines</td>
<td>432</td>
<td>424</td>
<td>0</td>
<td>2</td>
<td>141</td>
<td>0</td>
<td>1</td>
<td>3032</td>
</tr>
<tr>
<td></td>
<td>Otter board trawl</td>
<td>19</td>
<td>29</td>
<td>2</td>
<td>21</td>
<td>929</td>
<td>0</td>
<td>1</td>
<td>49300</td>
</tr>
<tr>
<td></td>
<td>Other</td>
<td>6</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>991</td>
<td>0</td>
<td>0</td>
<td>6136</td>
</tr>
<tr>
<td></td>
<td>Seine</td>
<td>268</td>
<td>595</td>
<td>0</td>
<td>58</td>
<td>64</td>
<td>0</td>
<td>14</td>
<td>845</td>
</tr>
<tr>
<td></td>
<td>Beam Trawl</td>
<td>24</td>
<td>11</td>
<td>2</td>
<td>907</td>
<td>0</td>
<td>50</td>
<td>6</td>
<td>983</td>
</tr>
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</table>

(Continued)
Table 7.2  (Continued).

<table>
<thead>
<tr>
<th>Country</th>
<th>Fleet and gears</th>
<th>COD</th>
<th>HAD</th>
<th>NEP</th>
<th>PLE Per mille</th>
<th>POK</th>
<th>SOL</th>
<th>WHG</th>
<th>ALL tons</th>
</tr>
</thead>
<tbody>
<tr>
<td>NETHERLANDS</td>
<td>Otter Board Trawl 70–99, 100–109, 110–119, &gt;4120, other</td>
<td>319</td>
<td>45</td>
<td>62</td>
<td>105</td>
<td>1</td>
<td>2</td>
<td>466</td>
<td>6758</td>
</tr>
<tr>
<td></td>
<td>Other</td>
<td>328</td>
<td>104</td>
<td>85</td>
<td>110</td>
<td>3</td>
<td>103</td>
<td>266</td>
<td>559</td>
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<tr>
<td></td>
<td>Beam Trawl 80–99, &gt;100, other</td>
<td>49</td>
<td>9</td>
<td>11</td>
<td>604</td>
<td>0</td>
<td>258</td>
<td>71</td>
<td>46490</td>
</tr>
<tr>
<td>SCOTLAND</td>
<td>Otter Board 70-99, 100-109, 110-119, &gt;120</td>
<td>120</td>
<td>576</td>
<td>81</td>
<td>11</td>
<td>53</td>
<td>0</td>
<td>159</td>
<td>94148</td>
</tr>
<tr>
<td></td>
<td>Other</td>
<td>116</td>
<td>464</td>
<td>204</td>
<td>8</td>
<td>33</td>
<td>0</td>
<td>175</td>
<td>4696</td>
</tr>
<tr>
<td></td>
<td>Seine 110–119, &gt;120</td>
<td>89</td>
<td>734</td>
<td>0</td>
<td>8</td>
<td>26</td>
<td>0</td>
<td>143</td>
<td>28418</td>
</tr>
<tr>
<td></td>
<td>Beam Trawl 80–99, &gt;100, other</td>
<td>23</td>
<td>21</td>
<td>1</td>
<td>913</td>
<td>0</td>
<td>31</td>
<td>12</td>
<td>6816</td>
</tr>
<tr>
<td>SWEDEN</td>
<td>Otter Board Trawl 70–99, 100–109, 110–119, &gt;120</td>
<td>151</td>
<td>333</td>
<td>5</td>
<td>1</td>
<td>507</td>
<td>0</td>
<td>3</td>
<td>2628</td>
</tr>
<tr>
<td>TOTAL</td>
<td>Catch weight in 1000 tons</td>
<td>42</td>
<td>105</td>
<td>14</td>
<td>68</td>
<td>106</td>
<td>16</td>
<td>40</td>
<td>689</td>
</tr>
</tbody>
</table>
indicating how important a species is. The choice of “decision weights” is a political one, as no objective method for setting the value appears to be available. Another set of political inputs is the “effort-reduction rates”, which determine how fast the effort of a fleet should be reduced. Furthermore, the model uses one more a priori (political) weighting factor, namely the so-called “relative importance of a species for a fleet” (the “fleet-target-factor”). Thus, the MTAC approach assumes that the “decision weights”, “effort-reduction rates” and “fleet-target-factors” are given beforehand.

There are no established rules for setting these inputs. The MTAC [75] includes three options for the effort reduction rates as examples. This illustrates some of the problems encountered when approaching effort-based regulation of fisheries: there is no established procedure for conversion of stock-based advice (catch quotas, TACs) to fleet-based advice (effort quotas). In general, there are no harvest control rules based on fleets and aimed at effort regulation.

An important part of current EU management principles is the relative stability of catch sharing between countries. TACs on key commercial species are divided between countries according to their historical rights. The implementation of the MTAC model for the North Sea considers seven stocks, and 78 fisheries (Table 7.2). The species are Cod, Haddock (HAD), Nephrops (NEP), Plaice (PLE), Saithe (POK), Sole (SOL), and Whiting (WHG). Table 7.2 also shows the total landings in 2002 [78]. The present version of MTAC does not account for historical rights. Somehow, this is not so apparent with the current use of MTAC: that is, for setting of total quotas only. However, should MTAC advice be used by managers for allocating catch or effort quotas to fleets, this becomes an important deficit of MTAC.

7.7.3.3 An Economic Interpretation of ICES ACFM Advice (EIAA)

Since 2002, the Subgroup of Economic Assessment (SGECA) of the STECF has used a model (the EIAA-model) to assess the economic consequences at the fleet segment level of the proposed quotas derived from the TACs proposed by ACFM [80].

This approach compares to the MTAC in the sense that the EIAA-model is based on the actual species compositions of the landings (three years historical average) of the fleet segments in question. Costs and earnings are calculated under the assumption that the fleet segments will take a constant share of each species irrespective of the size of the quotas.

As the model contains information about the total number of fishing days and the number of vessels in each fleet segment, the model can calculate the minimum required number of vessels to take the allocated quotas with respect to the assumption that the number of fishing days per vessel are increased to the maximum limit for vessels belonging to a specific segment. An estimate for overcapacity with given quotas can be derived.

The model also calculates an economically based estimate for overcapacity (based on the break-even principle known from business analyses). In this estimate the impact of prices and costs as well as catch rates are included.

The model does not include discards and assumes flexibility with respect to changes in catch compositions of the fleet segments equal to the changes in the overall quota composition pertinent for the fleet segment [74].
7.7.3.4 Advice and management problems in the knowledge base for effort regulation

The STECF WG on mixed fisheries did not directly address the questions on effort regulation, but looked only into the problems of developing TACs, that would account for the mixed nature of most fisheries, in particular the demersal fisheries. The model and the database used by this WG, the MTAC model, form a suitable framework for discussion of the problems for introducing effective effort regulation inherent in the knowledge base. Actually, as will be discussed below, the theoretical model used by the Mixed Fisheries Group of STECF could be used to calculate a maximum number of sea days.

The MTAC model, used by STECF to calculate TAC—advice, can also be used to calculate the effect of both fleet capacity reductions and limitations on sea-days. The STECF, however, did not use the model for those purposes, which here are given as a presentation of an opportunity, rather than a solution to the problem. Thus, the work by the STECF working group on mixed fisheries plays a central role in this discussion.

Limitations in the number of sea-days introduced in EU fisheries management in 2003–2005 are gear specific, as demonstrated in Table 7.3. The limitations are given for six gear groups, and reflect their catchability for cod and haddock: the smaller the number of sea-days, the higher the catchability of the gear for cod and haddock. Here should be noted the decreasing trend over the three years for all gear groups, except the large meshed trawl.

The actual maximum number of sea-days used in the management system (Table 7.3) appears not to be based on scientific advice from, for example, ICES or the STECF or any similar scientific body. However, the sea-day numbers are based on the management objective of reducing fishing mortality on selected stocks. If the target fishing mortality is given by the EU harvest control rules, the MTAC model could have been used to compute the corresponding maximum effort. Appendix 7.A demonstrates the (hypothetical)

<table>
<thead>
<tr>
<th>Grouping of fishing gears</th>
<th>(a) Demersal trawls, seines ≥ 100 mm&lt;sup&gt;a,c&lt;/sup&gt;</th>
<th>(b) Beam trawls ≥ 80 mm</th>
<th>(c) Static demersal nets</th>
<th>(d) Demersal long lines 80–99 mm&lt;sup&gt;b,c&lt;/sup&gt;</th>
<th>(e) Demersal trawls, seines 16–31 mm&lt;sup&gt;c&lt;/sup&gt;</th>
</tr>
</thead>
<tbody>
<tr>
<td>2003</td>
<td>9</td>
<td>15</td>
<td>16</td>
<td>19</td>
<td>25</td>
</tr>
<tr>
<td>2004</td>
<td>10</td>
<td>14</td>
<td>14</td>
<td>17</td>
<td>22</td>
</tr>
<tr>
<td>2005</td>
<td>9</td>
<td>13</td>
<td>13</td>
<td>16</td>
<td>21</td>
</tr>
</tbody>
</table>

<sup>a</sup> Skagerrak/Kattegat ≥ 90 mm
<sup>b</sup> Skagerak/Kattegat 70–89 mm
<sup>c</sup> Except beam trawls

procedure for setting the number of vessels and number of sea days with the aid of the MTAC model. The MTAC is actually used by the STECF to compute scenarios for the mixed fisheries’ TACs. The STECF did not present the calculations of maximum number of sea-days described in Appendix 7.A.

The MTAC quota proposals have, however, formed the basis for one of the scenarios carried out in the EIAP model, and, thus, the model, under certain assumptions, calculates estimates of overcapacity [83]. However, only the economic consequences for the current fleet capacity—and not the reduced capacity—are published.

One major benefit resulting from the use of sea-days, as compared to TACs, as the main tool for fisheries management in relation to demands for the knowledge base for the management system, is that sea-days will conserve resources even if stock sizes are poorly estimated. Nor does it require an exact knowledge of the discards, as is the case for properly set TACs. But management of fisheries by effort control (capacity and sea-days) raises a series of other problems in the knowledge base, including:

1. Allocation of effort between fleets (and countries);
2. Allocation of capacity between fleets (and countries);
3. Allocation of sea-days by fleet;
4. Relative stability in effort units to replace relative stability in catches;
5. Definition of fleet-based harvest control rules especially in relation to mixed fisheries (as illustrated by Table 7.3);
6. Definition of fleets and fisheries (for example, Table 7.2 compared to Table 7.3);
7. Definition of effort (sea-days, fishing days, trawling hours, HP days, etc.);
8. The interaction between management measures (license system, TAC, technical measures, maximum sea-days/month);
9. Compliance, enforcement (implementation errors);

Such problems are key in a multi-jurisdictional system with mixed fisheries, and where relative stability prevails as the central distributional approach of fisheries management.

Allocation of effort (capacity and sea-days) between fleets is a problem that corresponds to the current allocation of TACs between countries in the EU. The current sharing system is based on historical rights (relative stability). Rules for the establishment of historical rights in terms of capacity and sea-days could be introduced. Such fixed rules, however, may hamper the adaptation of fleets to a given resource situation, which is different from the historical situation. However, in any case, rules have to be agreed on. The use of the TAC rules for effort management (as exemplified by the MTAC model) somehow forfeit the benefits of effort control. The MTAC approach is an ad hoc method for solving the existing practical problems, but it should not become the long term solution.

With reference to (4), it is noted that most management measures have the same objective—namely to reduce the fishing pressure on over-exploited fish stocks. Therefore, they are all tested on their ability to recover stocks, and it becomes very difficult to separate the effects of the competing management measures. As an example, the reduction in vessel capacity produced by the MAGP essentially had the same effect with respect to fishing mortality as limitations on sea-days.
Relative stability—that is the stability of the fishing capacity and activity of countries—may or may not become the basic principle for allocations of fishing rights. An alternative would be to consider the fleets or fisheries the basic allocation unit, rather than the countries. Such a solution may be beneficial from a resource conservation point of view, but may not become feasible in the political world. Table 7.3 represents a step in this direction.

7.7.3.5 Implementation: Control and enforcement. Compliance

The definition of effort/fleet-based HCRs is the basis of any effort/fleet-based management regime. One possibility could be to use relative stability, perhaps combined with other instruments, to gradually adapt the fleets to new resource situations.

The definition of fleets and fisheries may have a great impact on actual management. Naturally, all vessel-owners will try to become members of the fleets with the fewest restrictions. For example, gear groups (a) and (e) in Table 7.3 are both demersal trawls, and they differ only with respect to the mesh size. Group (e), however, has a much higher allocation of sea days. The definition of the fleet concept is problematic in many respects, and represents perhaps the biggest problem. In that respect, although complex, the stock concept is easier to handle. Table 7.3 is to be considered a first ad hoc attempt to define fleets and fisheries for the North Sea.

The definition of effort may make a difference with respect to effort allocation. For Table 7.3, the definition “days absent from port” has been used. A “day present within the area and absent from port” is defined as one of the two options (member countries can choose between a or b): (a) the 24-hour period between 00:00 hours of a calendar day and 24:00 hours of the same calendar day or any part of such a period during which a vessel is present within the area defined in point 2 and absent from port; (b) any continuous period of 24 hours as recorded in the EU logbook during which a vessel is present within the area defined in point 2 and absent from port or any part of any such time period. One could imagine many other definitions. The present definition is adequate for legal purposes but may not be very suitable for linking effort with fishing mortality. “Trawling hours”, for example, may better reflect the fishing mortality created by a fishing operation than “days absent from port”.

Effort management by sea-day limitation is currently combined with all the traditional management tools. Therefore, the current implementation of effort regulation by sea-days does not remove the problems of those traditional management tools. Furthermore, separating the effect of effort regulation from the traditional regulations is almost impossible. There is no doubt that the current effort regulation by sea-days places extra stress on the fishing industry, and makes fishing less profitable in the short run. But in the long run, it may be beneficial for the stocks, because fishing is made so hard that investment decreases and dis-investment increases, thereby effectively reducing fishing mortality.

To achieve the full benefits from effort regulation it should replace some of the existing regulations. In theory, it should not be necessary to have both TACs and effort control, as they compete for the achievement of the same objective. The effort control should theoretically guarantee that catches do not exceed a given target. Thus, if one of the
measures could be fully implemented with full compliance of the industry and/or total enforcement, the other one would become superfluous. Somehow, the success criteria of the competing management tool should be a minimum of implementation errors. To that criteria should be added a regard for the profitability of the industry.

Compliance and enforcement play a crucial role for the implementation of regulations. One advantage of sea-days is that the rules are very simple (Table 7.3) and relatively easy to control with modern surveillance equipment. The simplicity indicated by Table 7.3, however, should be seen in the context of the problems discussed below. Table 7.3 does not contain the full legislation, of which certain articles are rather complicated.

7.8 CONCLUSIONS

A number of general characteristics of the knowledge base for different types of effort regulation systems presented in the examples can be summarised as follows in table form (Table 7.4):

The knowledge base of an effort regulation system is more complex in multi-jurisdictional international fisheries management systems than in national and unijurisdictional systems. Also, it is more complex for mixed fisheries systems compared to single species/stock fisheries.

Extensive problems associated with sharing common resources arise—for example, maintaining historical international relative stability. In general, it is easier for countries to agree to share catch in some proportion than to agree how to share out fishing effort. It will be necessary for politicians and management to have exact and detailed international

| Table 7.4 Characteristics of the knowledge base for the described types of effort regulation systems. |
|---------------------------------------------|------------------|------------------|------------------|------------------|------------------|------------------|------------------|------------------|
| Regulation Type                          | Capacity regulation by license | Activity regulation | TAC Transferable Rights | Decommission Progr. | Additional Technical Regulations | By-Catch & Discard Regulations |
| Case study                      | ITQ-catch | ITE-effort | ITQ-catch | ITE-effort | ITQ-catch | ITE-effort | ITQ-catch | ITE-effort | ITQ-catch | ITE-effort | ITQ-catch | ITE-effort |
| NPF Australia                      | Yes      | Indirect | No      | No      | Yes      | Yes      | Yes      | Yes      | Yes         |
| Faeroese Gadoid fisheries          | Yes      | Direct: Max sea days | No      | No      | Yes      | Yes      | Yes      | Yes      | Yes         |
| Danish Mussel fisheries           | Yes      | Indirect | (Yes)   | No      | No      | Yes      | Yes      | No       | No          |
| Mediterranean Hake fisheries      | Yes      | Indirect | No      | No      | No      | (Yes)   | Yes      | No       | No          |
| North Sea Demersal fisheries      | Yes      | Direct: Max sea days | Yes | No      | No      | Yes      | Yes      | Yes      | Yes         |

(Continued)
### Table 7.4 (Continued).

<table>
<thead>
<tr>
<th>Case study</th>
<th>Effort based management objective</th>
<th>Specific, quantitative based management objectives</th>
<th>HCRs / Reference Points (corresponding to specific management objective)</th>
<th>Universal, qualitative management objectives</th>
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</thead>
<tbody>
<tr>
<td>NPF Australia</td>
<td>Only capacity, not activity</td>
<td>Stock based, biological Fleet based, capacity</td>
<td>MSY transformed to $S_{MSY}$ and to $E_{MSY}$</td>
<td>Economic efficiency Ecological sustainable</td>
</tr>
<tr>
<td>Faeroese Gadoid fisheries</td>
<td>Only capacity, not activity</td>
<td>Stock based, biological Fleet based, capacity</td>
<td>SSB, F Ref. points transformed to effort levels</td>
<td>Economic sustainable Ecological sustainable</td>
</tr>
<tr>
<td>Danish Mussel fisheries</td>
<td>Only capacity, not activity</td>
<td>Fleet based, capacity</td>
<td>No</td>
<td>Economic sustainable Ecological sustainable</td>
</tr>
<tr>
<td>Mediterranean Hake fisheries</td>
<td>Only capacity, not activity</td>
<td>No</td>
<td>No</td>
<td>Overall fleet capacity Ecological sustainable</td>
</tr>
<tr>
<td>North Sea Demersal fisheries</td>
<td>Only capacity, not activity</td>
<td>Stock based, biological Fleet based, capacity</td>
<td>SSB, F Ref. points (TAE on top of that)</td>
<td>Economic sustainable Ecological sustainable</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Case study</th>
<th>Uni- or Multi-Jurisdictional System</th>
<th>Stakeholder Involvement</th>
<th>Type of Assessment Models</th>
<th>Data Level</th>
<th>Effort Unit definition</th>
<th>Performance/Utility Criteria defined</th>
</tr>
</thead>
<tbody>
<tr>
<td>NPF Australia</td>
<td>National</td>
<td>Yes</td>
<td>Biol. Prod., stocks Econ., fisheries</td>
<td>Data rich</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Faeroese Gadoid fisheries</td>
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<td>Yes</td>
<td>Biol. VPA, (stocks)</td>
<td>Data rich</td>
<td>Yes</td>
<td>Yes</td>
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<td>Danish Mussel fisheries</td>
<td>National</td>
<td>Yes</td>
<td>Stock indications Econ. survey</td>
<td>Data neutral</td>
<td>No</td>
<td>No</td>
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<tr>
<td>Mediterranean Hake fisheries</td>
<td>International</td>
<td>No</td>
<td>Limit. Biol. Eval. Limit. Econ. Fish.</td>
<td>Data poor</td>
<td>No</td>
<td>No</td>
</tr>
<tr>
<td>North Sea Demersal fisheries</td>
<td>International</td>
<td>No</td>
<td>Biol. VPA Stock Econ., fisheries</td>
<td>Data rich</td>
<td>Yes</td>
<td>Yes</td>
</tr>
</tbody>
</table>
Effort and capacity-based fisheries management

agreements on fisheries management and how to share the common resources. This will involve the formulation of precise management objectives and precise HCRs, as well as clear management strategies for the exploitation of resources.

Existing effort regulation systems are characterised only by the sparse formulation of specific and precise management objectives for proper input levels for implementation (especially precise activity levels, but also more precise efficient capacity levels), as well as precise quantitative HCRs. The definition of effort/fleet based HCRs provides the basis for any effort/fleet-based management regime. Fixed rules for the establishment of rights in terms of capacity and sea-days need to be agreed upon. Such rules may hamper the adaptation of fleets to a given resource situation and changes in that resource situation. Consequently, HCRs must consider both the technical interactions between fleets (exploitation and competition for the same resources by several fleets) and the biological interactions (fish predation and food competition) and changes in the resource situation in relation to mixed fisheries. A certain effort level for a mixed fisheries fleet will give conflicting effort levels by species in multi-species fisheries in relation to species-specific limit reference points for sustainability, and, consequently, some stocks will be at risk of being over-fished, and some under-fished, as only one stock can be fished on the single species optimal level. In mixed fisheries, it will be difficult to scale effort to harvest all target species and stocks in the fishery in the most economically optimal, efficient and biologically sustainable manner. Accordingly, HCRs must precisely specify the trade-off of reaching reference limit points for only one species or compromising between reference limit points for different species in the mixed fishery. In this respect, managers must also recognise that a management strategy can meet some objectives but conflict with others (which also can be a result of interaction between management measures), and multiple objectives accordingly have to be balanced and trade-offs decided upon in relation to the knowledge base.

To evaluate the success of different management options under an input regulation system, it has proved necessary to obtain precisely defined and quantitatively based biological, technological and economic performance and utility criteria, in relation to management objectives, as a part of the knowledge base, as well as know the sensitivity of those criteria.

Common units are essential to the knowledge base for making precise quantitatively based HCRs and to compare the effects of management strategies. Fleet-based HCRs (especially in relation to mixed fisheries) need to be defined. To do so, a precise definition and division of fleets (and fisheries) is necessary. The definition of fleets and fisheries is also essential to the knowledge base and may have a great impact on the actual management. Additionally, it is necessary to use a very precise and quantitatively based definition of a common unit of effort. The definition of effort makes a difference with respect to effort allocation. On the basis of a precise definition it is possible to allocate effort and capacity between fleets and countries. Further, to allocate the sea-days by fleet it is necessary to know and estimate the relative fishing power changes (efficiency, including technical efficiency) over time of the given fleet for a given species compared to other fleets.

In order to achieve relative stability in effort units, it is necessary to adjust fishing power by fleet and species and to use instruments that gradually adapt the fleets to
technological improvements and new resource situations. Accordingly, robust models for fishing power changes over time must be developed on the basis of good quality information—for example, precise data on partial catches, effort and catch rates by fleet. Consequently, it will be necessary to have, in the knowledge base for an input regulation system, systems to monitor, and tools to compensate for, unequal development in efficiency (effort creeping) over time between different national fleets and types of fleets. Even though this will be complicated, such models have been developed in several input regulation systems worldwide. However, most existing input regulation systems do not continuously and systematically monitor and make adjustments for effort creeping by technological improvements in recent years in management and management advice: as such, they do not take this important facet directly into account in fisheries management.

In some systems, it will be necessary to introduce relative stability in effort units to replace relative stability in catches—the complexities in scaling allowable effort to TACs must be solved, while also taking into consideration efficiency increases. In practice, no input management system is free of effort creep. Many input systems include continuous approaches to cutting capacity and effort. However, due to fleet efficiency changes over time the effective effort and capacity has very often not been reduced to the targeted level, nor has over-capacity been removed. This type of policy failure reduces the effectiveness of an input regulation system.

Property rights need to be considered in the context of the knowledge base for uni- and multi-jurisdictional fisheries management systems. Typically, resources do not have associated private property rights in effort regulation systems. In some input management systems the access to the fishery (licences) and/or the certain amount of activity given to a certain capacity are obtained as a private and transferable property right, which will be an object for trade and investment. Input-based properties are easier to define precisely, observe and control, and to confine. In output systems, the properties take the form of the resources and will be much more variable, very difficult to exactly define, observe and confine (poor definition of boundaries). Often resources will have very variable distribution and density patterns over time according to a long list of physical, biological and human impact factors. As a result, there is a variety of additional factors affecting property rights in such systems, and these will provoke weakness in control over the resources.

In input regulation systems, the resources will most often be public property under a transferable quota system; in contrast to an output based ITQ-management. This might be a very important aspect in management systems for many stakeholder interests besides those from the fishery, fishing industry and fisheries managers, and also important, when the resources are international property, where it might be more difficult to agree on transferring resources to private rather than international property.

In output management systems, it is necessary to have relatively precise knowledge and certainty about resource abundance: TAC-regulation depends on precise annual assessment and forecast estimates and advanced models to reach estimates. Uncertainty can lead to setting too high a TAC, followed by excessive pressure on stocks. Too low TACs can result in a missed opportunity for fishers to profit. Management through effort control has a similar problem. In order to be efficient, input regulation systems also need the target catch to be set to be able to manage stocks and to achieve maximum net return for
the fishery. Even though effort-based management is less dependent on precise yearly
stock assessment estimates and has a reduced need for annual assessment, it will still
need reasonably precise stock assessment and forecasts to set an appropriate initial effort
level and to respond to changes in efficiency. This can also be obtained by judging the
removal rates over a series of years generated by recent fishing effort levels. However,
input management can result in economic inefficiency where too much capacity expend-
ing too much effort to catch the limited resources results in lower net return than could
otherwise be achieved. Consequently, the full economic potential of the fishery is not
realised. In fisheries systems where stock sizes are poorly known or estimated the effort
regulation will be able to provide conservation of resources. It has been recognised that
input management is relatively more effective than output management when variability
in catch rates is less than the variability in stock levels. In this respect the lack of precise
knowledge about stock–recruitment relationships complicates evaluation of the efficiency
of both output and input-based management.

Fleet-based fisheries assessment models and sufficient biological, economic and tech-
nological data to support these, have only been applied and used in management advice to
a limited extent in input management systems worldwide. Consequently, scientific advice
does not support these management systems fully by delivering quantitative consequence
scenarios for biological, economic and social consequences of different management
options.

Effort regulation often has the explicit purpose of reducing discards. Typically, such
systems will have the effect of reducing the incentives to discard (see the Introduction).
In the knowledge base, there is not as high a need for extensive and precise discard data
as there would be in output regulation systems, where it is a priority to estimate total
catch.

Over-capacity has been shown to be a problem in all input regulation systems as it
results in high pressure on the resources and non-compliance with regulations. Despite
several examples of approaches to reduce the capacity, these were only partly successful
within various input regulation systems, due to the technological efficiency increase over
time as well as substitution of non-regulated input.

The knowledge basis must contain necessary information about expected reaction pat-
terns of fishermen to different types of regulations as well factors affecting the likelihood
of compliance. Control of one or more inputs provides an immediate incentive for fishers
to substitute uncontrolled input, which is the case in some systems. As input regulation
regimes provide no ownership of the fisheries resources there can, similarly to output
regulation systems, be an incentive for heightened competition for the catch—this will
often mean more input and reduced economic efficiency. Compliance seems to have an
tendency to be greater with those systems where there is a strong stakeholder involvement
in advisory processes, as well as stakeholder participation in the management bodies of
the decision-making process. With respect to compliance and enforcement under effort
regulation, control under this system is simpler than in an output regulation system
because it is easier to observe activity of vessels than to monitor catch and output. The
improvement in information resulting from VMS registration of activity is an important
means to achieve that in many systems.
APPENDIX 7.A: INTRODUCTION TO THE “MTAC MODEL” AND ITS USE FOR CALCULATION OF MAXIMUM NUMBER OF SEA DAYS

7.A.1 The EU implementation of the precautionary approach of fisheries management

The mathematical expression for the HCR of ICES for single species assessment is:

\[
F_{HCR}(y + 2) = \begin{cases} 
0 & \text{if } SSB(y) \leq B_{lim} \\
\frac{SSB(y) - B_{lim}}{B_{pa} - B_{lim}} F_{pa} & \text{if } B_{lim} \leq SSB(y) \leq B_{pa} \\
F_{pa} & \text{if } SSB(y) > B_{pa}
\end{cases}
\]  

(7.A.1)

Present ICES advice ignores that (almost) all fish are caught in mixed fisheries. In “mixed fishery” a quota on one species has influence on the catch of all the other species caught together with the quota-species.

7.A.2 Mean number of sea days

Effort of a single vessel can be measured as the number of “sea-days” (or days absent from port) in a month. The effort of a fleet (a group of similar vessel, see definition Section 7.7, is the sum over vessels. Let \(N_{oV_{Fl}}(m)\) be the number of vessels in fleet “Fl” in the month in question (\(m\)). Then the total effort in month “\(m\)” becomes the sum over individual vessels (index: \(Vs\))

\[
E_{Fl}(m) = \sum_{Vs=1}^{N_{oV_{Fl}}(m)} \text{SeaDays}(V_{S,m}) \times \text{VesselSize}(V_{S}).
\]  

(7.A.2)

“VesselSize” can be measured as horsepower or tonnage, according to the MAGP. Equation (7.A.2) can be rewritten

\[
E_{Fl}(m) = N_{oV_{Fl}}(m) \times \bar{D}_{Fl}(m) \times \overline{S_{Fl}}(m)^* 
\times \left[ \frac{1}{N_{oV_{Fl}}(m)} \sum_{Vs=1}^{N_{oV_{Fl}}(m)} \text{SeaDays}(V_{S,m}) \times \text{VesselSize}(V_{S}) \right].
\]  

(7.A.3)

Here \(\bar{D}_{Fl}(M)\) and \(\overline{S_{Fl}}\) indicate mean number of days per months and mean vessel size. If the individual vessels in the fleet do not deviate from the mean, the last term (in squared brackets) of Eq. (7.A.3) becomes 1 and Eq. (7.A.2) reduces to

\[
E_{Fl}(m) = N_{oV_{Fl}}(m) \times \bar{D}_{Fl}(m) \times \overline{S_{Fl}}(m).
\]  

(7.A.4)

If we assume the vessel size to be independent of months, and define the effort unit to be the effort of one average vessel (\(\overline{S_{Fl}}(m) = 1\)) then Eq. (7.A.4) reduces to

\[
E_{Fl}(m) = N_{oV_{Fl}}(m) \times \bar{D}_{Fl}(m).
\]  

(7.A.5)
The annual effort becomes the sum over months

\[ E_{Fl} = \sum_{m=1}^{12} \text{NoV}_{Fl}(m) \cdot \overline{D M}_{Fl}(m) \]  

(7.A.6)

which can be approximated by the mean number of vessel per month \( \overline{\text{NoV}}_{Fl} \) times the mean number of sea days per month \( \overline{D M}_{Fl} \).

\[ E_{Fl} = \overline{\text{NoV}}_{Fl} \cdot \overline{D M}_{Fl}. \]  

(7.A.7)

Equation (7.A.7) represents a definition of effort which is suitable for both types of effort regulation.

The MAGP aims at reducing the NOV and the limitation of sea-days aims at reducing the DM.

### 7.A.3 The STECF approach to mixed fisheries TACs

The MTAC approach is to minimize the sum of squares of deviations (SSD) between the target fishing mortalities (defined by ICES) and the fishing mortalities advised by the fisheries management.

\[ \text{SSD} = \sum_{Sp} \sum_{a} (F(Sp, a) - F_{PA}(Sp, a))^2 \]  

(7.A.8)

to be minimized.

“Sp” is index of “Species”, “a” is index of age group. The \( F \) in Equation (7.A.8) refers to the (future) TAC-year, often “next year”. The target fishing mortality may not always be the precautionary approach \( F \). Replacing \( F_{PA} \) in Eq. (7.A.8) with the more imprecise concept \( F_{Target} \) would make the equation general. The \( F_{Target} \) is usually expressed as a proportion of the status quo \( F \), the \( F \) of the most recent data year (or mean over some recent years). The minimization of SSD is an approach similar to meeting the multi-species inequalities. Meeting the multi-species inequality by species may have different priority for the managers.

This can be accounted for by introduction of the so-called “decision weights \( \theta_{Sp} \)”. These are species specific numbers indicating how important a species is. The sum is 1. The SSD now takes the form.

\[ \text{SSD} = \sum_{Sp} \sum_{a} \theta_{Sp} (F(Sp, a) - F_{PA}(Sp, a))^2 \]  

(7.A.9)

The choice of \( \theta_{Sp} \) is a political one, as no objective method for setting the \( \theta_{Sp} \) value appears to be available. The higher \( \theta_{Sp} \) is, the more emphasis is placed on species \( Sp \). In the extreme \( \theta_{Sp} = 1 \) and other \( \theta \)'s = 0, only species \( Sp \) is cared for.

Each \( F \) is composed of the fleet components

\[ F(Sp, a) = \sum_{Fl} F(Sp, Fl, a), \]
so SSD can be written as

\[
SSD = \sum_{Sp} \theta_{Sp} \left( \sum_{a} \sum_{Fl} F(Sp, Fl, a) - F_{PA}(Sp, a) \right)^2.
\]  

(7.A.10)

\(F(Sp, Fl, a)\) is usually expressed relative to the “current” \(F\) value, or the “Status Quo” value of \(F\), which means the most recently observed \(F\). \(F(Sp, Fl, a) = \max\{0, X_{Sp, Fl}\} \cdot F_{SQ}(Sp, Fl, a)\). The maximum function is used to secure that the fishing mortality will not get negative. This is required in the expressions where \(F\) is calculated.

\[
SSD = \sum_{Sp} \theta_{Sp} \left[ \sum_{a} \sum_{Fl} \max\{0, X_{Sp, Fl}\} \cdot F_{SQ}(Sp, Fl, a) - F_{PA}(Sp, a) \right]^2.
\]  

(7.A.11)

This SSD is a function of a matrix of \(X\)’es, and there is no unique value which minimise the SSD. But if one assume the relative values of the \(X\)’es to be given as constants, that is \(X_{Sp, Fl} = \alpha P_{Sp, Fl}\) where the \(p\)’s are constants and \(\alpha\) is a variable, then SSD is a function of only one variable, \(\alpha\), and there is a unique value which minimizes the SSD. The \(p\)’s within a species sum to 1.0 over fleets:

\[
SSD(\alpha) = \sum_{Sp} \theta_{Sp} \left[ \sum_{a} \sum_{Fl} \max\{0, \alpha P_{Sp, Fl}\} \cdot F_{SQ}(Sp, Fl, a) - F_{PA}(Sp, a) \right]^2.
\]  

(7.A.12)

The coefficients \(p\), are in the present context called the “effort-reduction rates” as they determine how fast the effort of a fleet should be reduced. Usually, this is about over-fishing, and consequently the term “effort reduction” is adequate. As for the \(\theta\)’s (decisions weights) the choice of \(p\)’s may be political ones. However, the STECF Working Group on Mixed Fisheries also suggested some “objective methods” to calculate \(p\). The choices of an “objective method”, however, are subjective, and the WG suggested three objective methods.

Furthermore, the MTAC model uses one more a priory coefficient, namely the so-called “relative importance of species \(Sp\) for fleet \(Fl\) (the “fleet-target-factor”) which sums to one over species for each fleet, so that the SSD reads:

\[
SSD(\alpha) = \sum_{Sp} \theta_{Sp} \left[ \sum_{a} \sum_{Fl} g_{Sp, Fl} \cdot \max\{0, \alpha P_{Sp, Fl}\} \cdot F_{SQ}(Sp, Fl, a) - F_{PA}(Sp, a) \right]^2.
\]  

(7.A.13)

Thus, the MTAC approach assumes that the coefficients \(\theta\), \(p\) and \(g\) are given beforehand.

The MTAC [75] suggests three options for the effort reduction coefficients (\(C\) is number caught in age group “\(a\)” and \(W\) is the body weight) and two options for \(g\) (the “fleet target factor”). The “decision weights” \(\theta\), are given arbitrary high values to cod.

Table A.1 illustrates some of the problems encountered when approaching effort based regulation of fisheries. There is no established procedure for conversion of stock-based
Table A.1  A priori coefficient required to run the MTAC model.

<table>
<thead>
<tr>
<th>Coeff./Option</th>
<th>(1) Equal weight</th>
<th>(2) Importance within the fleet, measured as relative weight of catch</th>
<th>(3) Importance within the species, across fleets, measured as relative weight of catch.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Effort-reduction rates, $p$</td>
<td>$p_{\text{Equal}}^{Sp,Fl} = 1$</td>
<td>$p_{\text{Within}}^{Sp,Fl} = \frac{\sum a C_{Fl,Sp,a} \times W_{Fl,Sp,a}}{\sum_{i(\text{species})} a \sum C_{Fl,i,a} \times W_{Fl,i}}$</td>
<td>$p_{\text{Across}}^{Sp,Fl} = \frac{\sum a C_{Fl,Sp,a} \times W_{Fl,Sp,a}}{\sum_{m(\text{fleet})} a \sum C_{m,Sp,a} \times W_{m,Sp}}$</td>
</tr>
<tr>
<td>Relative importance Sp for Fl: “target-factor”, $g$</td>
<td>$g_{\text{Equal}}^{Sp,Fl} = 1$</td>
<td>$g_{\text{Within}}^{Sp,Fl} = g_{\text{Within}}^{Sp,Fl}$</td>
<td>Not applicable</td>
</tr>
</tbody>
</table>

For $\theta$ is used the same value for all species except for cod, which has got assigned a value 20 times as big (We suppose these were more or less arbitrary choices)

advice (catch quotas, TAC) to fleet based advice (Effort quotas). In general, there are no harvest control rules based on fleets and aiming at effort regulation.

For a given value of $\alpha$ MTAC determines the combined fleet-effort-reduction factor by

$$ Fac_{Fl}(\alpha) = \sum_{Sp} \theta_{Sp} \cdot g_{Sp,Fl} \cdot \max\{0, \alpha \cdot p_{Sp,Fl}\} \quad (7.14) $$

MTAC starts the calculation of TAC by computing the total fishing mortality

$$ F(Sp, a) = \theta_{Sp} \sum_{Fl} g_{Sp,Fl} \cdot \max\{0, \alpha \cdot p_{Sp,Fl}\} \cdot F_{S_{Q}}(Sp, Fl, a) \quad (7.15) $$

where the partial fleet fishing mortality, $F_{S_{Q}}(Sp, Fl)$, is calculated by

$$ F_{S_{Q}}(Sp, Fl) = F_{S_{Q}}^{ICES}(Sp) \cdot \frac{C_{Fl,Sp,a}}{\sum_{k(fleet)} C_{k,Sp,a}} \quad (7.16) $$

Superscript “ICES” means that this value is delivered by an ICES assessment working group, which used some sort of cohort analysis for that purpose. From the $F$ combined with the stock numbers estimated by the ICES WG the TAC is computed by the catch equation.

$$ TAC_{Mixed\ Fishery}^{Sp} = \sum_{a} \sum_{Fl} F(Sp, Fl) \cdot N_{Sp,a}^{ICES} \cdot W_{Fl,Sp,a} \cdot \frac{1 - \exp(-Z_{r,Sp,a}^{ICES})}{Z_{Sp,a}^{ICES}} \quad (7.17) $$

where $N$ and $Z$ are the stock numbers and total mortality as estimated by the ICES WG.
7.A.4 Using the MTAC Model to calculate the number of sea-days

The derivations below are those of the current authors, not the STECF WG on mixed fisheries. If we write the status quo fishing mortality, $F_{SQ}$, as the product of catchability $Q$ and status quo effort, $F_{SQ}(Sp, Fl, a) = Q_{Sp,Fl,a} * E_{SQ}(Fl)$ (7.A.18)

then SSD reads:

$$SSD(\alpha) = \sum_{Sp} \theta_{Sp} \sum_{a} \left( \sum_{Fl} g_{Sp,Fl} \cdot \max\{0, \alpha \cdot p_{Sp,Fl}\} \cdot Q_{Sp,Fl,a} \cdot E_{SQ}(Fl) - F_{PA}(Sp, a) \right)^2$$ (7.A.19)

The effort of Fleet $Fl$ as a function of $\alpha$ is:

$$E_{Fl}(\alpha) = \sum_{Sp} \theta_{Sp} \cdot g_{Sp,Fl} \cdot \max\{0, \alpha \cdot p_{Sp,Fl}\} \cdot E_{SQ}(Fl)$$ (7.A.20)

The effort will usually be the annual effort, but it could be the effort of any time period, or the average monthly effort. $E$ could be the total number of sea-days by all vessels in the fleet. Effort can be measured as sea-days per month multiplied by the number of vessels in the fleet, $NoV_{Fl}$, (as introduced in 7.A.2) and we get the expression.

$$E_{Fl}(\alpha) = \sum_{Sp} \theta_{Sp} \cdot g_{Sp,Fl} \cdot \max\{0, \alpha \cdot p_{Sp,Fl}\} \cdot 12 \cdot DM_{SQ,Fl} \cdot NoV_{Fl}$$ (7.A.21)

$DM_{SQ,Fl}$ is the average number of sea days observed the most recent data year, the status quo value. $DM_{Fl}(\alpha)$ is the number of sea-days derived from $DM_{SQ,Fl}$, $\theta$, $p$ and $g$.

The MTAC is actually used by the STECF to compute the mixed fisheries TAC’s. The STECF did not present the calculations of max number of sea-days.

As $F$ is a function of the number of vessels (Eq 7.A.18 and Eq. 7.A.21), Eq 7.A.21 can be used to predict the short and long term effect of reductions in capacity in a mixed fishery.

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Section 2

ISSUES RELEVANT TO THE EUROPEAN LEVEL
Chapter 8

Fisheries Policy-Making: Production and Use of Knowledge

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8.1 INTRODUCTION

The institutional set-up for production and use of knowledge in relation to fisheries management in Europe is complex and involves a number of actors. The two most prominent actors in the system are the International Council for the Exploration of the Sea (ICES) and the European Union (EU, Union). ICES is the intergovernmental agency that—as the centre of a system that provides management advice based on biological scientific knowledge—coordinates marine research activities in the North Atlantic. The EU is the main consumer/user of this scientific knowledge as a part of its decision-making process relating to the management framework and instruments of the Common Fisheries Policy (CFP). The CFP is arguably one of the most science-dependent common policies in the portfolio of the EU.

The CFP is currently undergoing reform and a major step of the ongoing process was taken with the adoption of a new basic regulatory framework, which entered into force 1 January 2003 [1]. The present and ongoing reform of the CFP is closely linked to the fact that EU fisheries management has fallen short of delivering the desired results of fisheries management since the first basic regulation for a common fisheries policy was agreed upon in 1983. A large number of fish stocks are currently outside what have been defined as safe biological limits; and much of the catching industry is in a permanent state of crisis, due to continuing overcapacity and low quotas. This is described in the Commission’s Green Paper—a comprehensive preparatory document relating to the reform process [2]. The precarious situation of the cod stocks and the commercial whitefish sector in the North Sea has in recent years manifested itself as the prime example of this crisis.

Attempts at reforming the system that provides scientific and technical advice to the Union is also high on the agenda, and, in 2003, the Commission published a Communication on the issue [3]. Deficiencies relating to fisheries science and the provision of advice have, from various sides, including from outside the Commission, been identified to be contributing to the CFP’s problems [4].

This chapter provides a basic introduction to the present decision-making system, how scientific knowledge and other types of knowledge feed into it, the problems inherent in
the system and the contours of the reform of the system that delivers scientific advice and other types of knowledge.

8.2 THE ICES–EU SYSTEM

The CFP aims at the sustainable exploitation and equitable distribution of a resource that is constantly changing in complex ways in response to human exploitation as well as to natural factors. The setup of the present system means that the EU is in perpetual need of accurate assessments of the state of the fish stocks in order to apply the appropriate management measures. The CFP’s basic regulation states that the decision-making process shall be “based on sound scientific advice which delivers timely results” [1]. The biological element of this scientific advice is delivered mainly by ICES but the Union is also obliged to seek information and input from other sources. This follows from a requirement to involve stakeholders at all stages of policy-making [1].

The process that transforms ICES’ scientifically-based advice, and other inputs of knowledge into EU fisheries management measures, is not static or uniform. It is hardly possible to arrive at an objective answer as to what input is appropriate in relation to a specific measure. The discussion of the balance between inputs from different stakeholders in relation to the CFP’s decision-making process has consequently been one of the main issues of controversy of the present reform of the system. Biologically based scientific advice might be interpreted as pivotal in some cases—for example, on the question of total allowable catches (TACs); and equally insignificant in other cases—for example, on questions of compensation, where economic information is more relevant. The interpretation will to a large extent depend on who is asked. As a result of this, the actors involved in providing the knowledge base vary from measure to measure and over time.

The formal decision-making process of the EU varies also depending on the issue in question. Whereas some degree of involvement of the Commission and the Council is usually given, the involvement of the Parliament and various committees varies. Figure 8.1 provides a schematic overview of the advice and management system, with a focus on ICES and the EU.

8.2.1 International Council for the Exploration of the Sea

ICES delivers scientific advice on fisheries management to the EU, among other clients. The EU is ICES’ largest client and ICES is likewise the most prominent provider of scientific advice to the EU in the area of fisheries. ICES provides advice on fish stocks in the North-East Atlantic, where some of the Union’s most important fisheries takes place. ICES’ assessments are based on both data from commercial fishing vessels (fisheries-dependent data) and data collected through other means, for instance trawl surveys conducted by research vessels (fisheries-independent data).
ICES has 19 member states¹, and can be understood as a forum where the national laboratories of the ICES member states pool their resources. The basic unit of the ICES system is the individual scientists, who are employed in the different national laboratories and institutes of the ICES member states. The network consists of approximately 1600 marine scientists, mainly biologists. The national fisheries institutes are funded by the member states, but, specifically in relation to the EU member states, the Commission funds an increasing amount of activities within the national institutes. It is, however, questionable how much this increasing contribution actually increases the overall budgets of the institutes seen in the light of national budget cuts. ICES’ own budget does not cover more than coordination activities and it is therefore of crucial importance to ICES that the national institutes have sufficient funding. The work, which is necessary for ICES to carry out its tasks, is coordinated through a system of committees and working/study groups. Some of the groups are maintained in cooperation with other international fisheries organisations.

There are more than 100 working/study groups dedicated to different themes covering all aspects of the marine ecosystem. Many of these are charged with specific species or specific areas; others relate to methods or other issues such as by-catch. The members of these groups are nationally appointed experts from the member states’ fisheries institutes and universities, etc. These working groups submit reports to either: the Advisory Committee on Fishery Management (ACFM), which advises on the state of living marine

¹ The ICES member states are Belgium, Canada, Denmark, Estonia, Finland, France, Germany, Iceland, Ireland, Latvia, the Netherlands, Norway, Poland, Portugal, Russia, Spain, Sweden, the United Kingdom, and the United States of America.
resources; the Advisory Committee on the Marine Environment (ACME), which advises on issues such as marine pollution; or the Advisory Committee on Ecosystems (ACE), which advises on ecosystem dynamics.

These are the three committees responsible for delivering advice to the clients. Besides this the Committees are charged with overseeing the work of a number of working/study groups where the basis for the advice is developed. The advice is—after having been compiled and reviewed by the relevant advisory committee—forwarded to the client. The EU receives the advice from the ACFM, and it is on this advice that proposed TACs for the key commercial species are based. That the EU is, in economic terms, ICES’ largest client means that this relationship is particularly important to ICES. As a result, the organisation has been responsive to the wishes of the EU (see section 8.4).

8.2.2 European Union

The main actors in relation to the direct formulation of the Common Fisheries Policy—and therefore also the main consumers of knowledge—are the European Commission’s Directorate-General for Fisheries and Maritime Affairs (DG Fisheries) and the legislating body of the EU’s Agriculture and Fisheries Council, where the Member States are each represented by their Minister responsible for fisheries. A less prominent role is played by the European Parliament (EP) and its Committee for Fisheries, which has, if nothing else is stated, the right to be heard on matters relating to the CFP. The Court of Justice of the European Communities will not be dealt with in this context, as it does not play an active part in the decision-making process. It should, however, not be forgotten that some of its rulings have influenced EU fisheries management significantly by clarifying the valid interpretation of EU legislation.

The corner stone of the CFP’s conservation policy is limitation of catches by the setting of annual TACs for single species in defined geographical areas (the ICES-areas). TACs are set for most of the commercially important species. The TACs are subsequently distributed in predetermined shares between the Member States, which, in essence, means that a Member State’s national allocation can only increase if the overall TAC increases. This core principle is called ‘relative stability’. The quota shares were originally calculated on the basis of a combination of (1) historic catches; (2) special provisions for coastal communities, which were heavily dependent on fishing; and (3) compensation for jurisdictional losses in catches in the waters of third countries, which resulted from the establishment of 200 nautical mile exclusive economic zones (EEZ) in the mid-1970s [5]. The other core principle of the CFP is ‘equal access’, which means that the combined sea area of the member states’ EEZs is treated as a common European sea area (sometimes referred to as the ‘Community pond’), although with special conditions applying within the 12 nm limit. The TAC system has traditionally been supplemented by a variety of technical measures, which are directed mainly at preventing the (by-)catching of juvenile fish or non-target species. Technical measures include provisions for minimum mesh sizes, minimum landing sizes, rules as to what fishing gear can be used and where, seasonal bans on fishing, limitations on days-at-sea for vessels, etc. In connection with the application of the reformed CFP from 1 January 2003, renewed focus has been put on
the limitation of fishing effort as a way to limit pressure on stocks—particularly within the framework of multi-annual recovery or management plans [1].

Whereas the conservation component of the CFP is highly dependent on biological advice concerning stock levels, this is not the case, to the same extent, for other components of the CFP. Biological advice is not generally considered in the decision-making process relating to the structural policy of which the main instrument is the Financial Instrument for Fisheries Guidance (FIFG). This relative lack of attention to biological knowledge is to some extent paradoxical since this component of the CFP has traditionally had an immense impact on the fishing industry and, indirectly, also on the state of the stocks. This is because the CFP’s structural policy has until recently, at least in part, contributed to the development of overcapacity in the European fleet: overcapacity has been identified as one of the main issues to address in order to achieve sustainable fisheries. This exclusion of biological knowledge has contributed to a relative detachment between the conservation component and other components of the CFP, meaning that different parts of the CFP have, in reality, been pulling (partly) in different directions. In the same way, it could be argued that there should be more focus on socio-economics within the advice related to the conservation component of the policy, since this is basically about managing fisheries, rather than fish. These issues are to some extent taken into consideration by the ongoing reform of the CFP.

8.2.2.1 The Commission / DG Fisheries

Advice from ICES is, in the first instance, received by DG Fisheries. The tasks of this division of the European Commission are to develop and propose measures to advance the objectives of the Union; and to implement those measures under the executive powers delegated to it by the Council [6]. To be able to perform these tasks smoothly and efficiently, an appropriate knowledge base is of pivotal importance. But, as mentioned earlier, the sources of appropriate knowledge depend upon the issue in question and the point-of-view of those seeking the knowledge. DG Fisheries officials perceive, according to Lequesne [7], themselves as guardians of expertise, especially biological expertise, as opposed to governments, which are vulnerable to lobbying efforts from the industry.

The Commissioners are supposed to act on behalf of the Community, and it is, therefore, a legal requirement that they remain independent and not take instructions from either governments or other bodies. In practice, Member States actively try to place their Commissioner in charge of their preferred portfolio and the requirement for independence and neutrality is interpreted with some degree of flexibility [8]. Commissioners are, from time to time, accused of taking instructions from and blatantly protecting the interests of the Member State from which they come. This is generally not considered acceptable.\(^2\)

\(^2\)An incident during the reform year of 2002 is a good example of the, at times, less than neutral position of Commissioners. On national television, the Spanish Fisheries Minister, Miguel Aries Cañete, declared on 25 April that Spain and other Member States opposing the reform (as it was outlined at the time) had instructed their Commissioners to obstruct the reform. The Spanish Minister later withdrew his statements and Spain suffered no formal consequences for the affair. However, the affair received substantial negative publicity, which shows that there are limits to the flexibility of the interpretation of neutrality. This political attempt to use Commissioners to obstruct a European policy was clearly considered out of line.
The Commission has the right to propose new legislation in the area of fisheries policy, as in most EU policy areas. It is DG Fisheries that annually drafts proposals for the TACs for the following year. These proposals are usually ready by the beginning of December. As a part of their preparation, DG Fisheries acquires knowledge from various sources. Their in-house scientific capacity is very limited and counts only nine persons as of April 2002. Thus, scientific knowledge must be provided from outside the organisation [3]. The sources of scientific knowledge, and knowledge in a broader sense, include, besides ICES and other international organisations, several committees or committee-like structures. In connection with the setting of TACs the Scientific, Technical and Economic Committee for Fisheries (STECF) is of particular importance.

Other sources of knowledge input can also be included in the process but this will more often be in relation to more fundamental framework decisions like the new basic regulation. On a number of occasions the Commission has engaged in dialogue with stakeholders through the use of public hearings, where concerned interests are encouraged to give their input. In its most simple form stakeholder involvement entails inviting opinions through DG Fisheries’ website. However, in relation to the preparation of the Green Paper on reform, the consultation process involved elements such as sending out questionnaires to 350 groups with an interest in fisheries, a large number of regional meetings in the various Member States, and a concluding conference attended by 400 people in Brussels. In addition, the EU, represented by the Commission, funds a large number of different research projects through its research framework programmes: the sixth and current programme runs from 2002 to 2006. This publication is funded under priority 8 of the Sixth Framework Programme—Integrating and strengthening the European Research Area: Policy-Oriented Research. The ability of the Commission to formulate programme areas, thematic priorities and calls for proposals, which eventually result in research results, enables it to guide the European research agenda according to its needs. Although the Commission cannot determine the results of such research, it can, to some extent, decide upon its’ focus. Furthermore, Member States are legally required to obtain and provide certain information on biology, economy and social issues relating to fisheries.

The datasets, which the Member States are (or in coming years will be) legally required to collect and provide, are outlined in a body of regulations. The Council’s basic framework regulation on collection and management of data states that “[t]o conduct the scientific evaluations needed for the common fisheries policy [. . .], complete data must be collected on the biology of the fish stocks, on the fleets and their activities and on economic and social issues” [9]. The consequences of this, in terms of specific data requirements and confidence levels, are subsequently outlined in detail in a Commission regulation (as amended by a later regulation [10]), which groups data in three modules: (1) module of evaluation of inputs: fishing capacities and fishing effort; (2) module of evaluation and of sampling of catches and landings; and (3) module of evaluation of the economic situation of the sector [11]. The fact that the plans for data gathering are national responsibilities, while fisheries resources are shared, is a problem. The Commission is now, therefore, setting up coordination meetings to facilitate cooperation in this matter.
The progressive implementation and adaptation of this regulatory framework\(^3\) strengthens the knowledge base for EU fisheries management by contributing to the improvement of the data, which policy-makers, experts and stakeholders rely on in relation to the CFP’s decision-making process.

When DG Fisheries has received the required information from the relevant sources, including input from other relevant DGs, the responsible directorate\(^4\) in DG Fisheries finishes drafting the proposal, which is then passed upwards through a number of stages. The final internal destination of any Commission draft proposal is the ‘College of Commissioners’, which can accept the proposal, reject it, refer it back for re-drafting or decide not to take any decision whatsoever. The College of Commissioners consists presently of one Commissioner from each of the 25 member states and takes decisions by simple majority through secret voting.

The proposal from DG Fisheries (and eventually the Commission) on TACs is often relatively similar to that recommended by STECF/ICES. However, this is not always the case. The Commission is not only concerned with long-term biological sustainability of fish stocks, but also takes into consideration both the short and longer-term socio-economic costs of TAC-related decisions. The step from STECF to the Commission—or perhaps especially from DG Fisheries to the College of Commissioners—constitutes a step away from almost purely biological scientific considerations and towards a phase where politics and other objectives are increasingly taken into consideration. Consequently, the proposal on TACs, which passes from the Commission to the Council, often involves higher TACs than those that would follow from the scientists’ biological advice, even though DG Fisheries perceives itself, to a considerable extent, to be the guardian of exactly that.

Central to the system that provides knowledge from within the EU system stand a number of committees with varying responsibilities and relationships with DG Fisheries.

**Scientific, Technical and Economic Committee for Fisheries**

The most important committee in relation to scientific advice is the Scientific, Technical and Economic Committee for Fisheries (STECF). This Committee is provided for by the basic regulation, which states that the Committee “shall be consulted at regular intervals on matters pertaining to the conservation and management of living aquatic resources, including biological, economic, environmental, social and technical considerations” [1]. This advisory expert Committee consists of 30 to 35 scientists, mainly from the fields of marine biology, marine ecology, fisheries science, nature conservation, population dynamics, statistics, fishing gear technology, aquaculture, and the economics of fisheries and aquaculture, who are appointed for a renewable period of three years [12]. The Committee (under its former name—the Scientific and Technical Committee for Fisheries)

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\(^3\) Scoreboards and guidelines are also connected to the body of regulations. These are intended to improve the provision of data from the Member States and to increase its uniformity.

\(^4\) DG Fisheries consists—as of April 2005—of 5 directorates relating to different aspects of the CFP namely: conservation policy, external policy and markets, structural policy, control and enforcement, and, finally, resources and relations with stakeholders. See DG Fisheries organigramme: http://europa.eu.int/comm/dgs/fisheries/organig_en.pdf [Accessed 15 April 2005].
was provided for by the first basic regulation of the CFP from 1983; it was not until 1993 that the Committee composition was altered to include economists [5].

STECF meets in plenary twice a year and advises DG Fisheries on matters where scientific skills are central. This advisory Committee can, in collaboration with DG Fisheries, form internal sub-groups/working groups, which, in some cases, include experts from outside the STECF and even the EU Member States [3]. STECF has some latitude with regard to the advice it produces as it can, by its own accord, draw attention to the same matters on which it is regularly consulted [12]. STECF relies, to a large extent, on the same expertise as ICES, namely that emanating from the national fisheries laboratories. This means that there is a considerable overlap between members of the STECF, including its subgroups, and the various ICES groups and committees. This leads, according to the Commission, to repetitive work for some experts [3].

STECF bases the advice on TACs and other issues, which it gives to DG Fisheries, predominantly on the findings produced by ICES, but also on those of other scientific advisory organisations (particularly regional organisations, which will be dealt with later in this chapter). STECF is, in other words, consulted to review the yearly scientific advice from ACFM, as well as other advice coming from ICES or other sources. The advice that STECF submits to DG Fisheries rarely differs substantially from that of the original from ICES. However, STECF adds to the knowledge base around the advice by conducting evaluations for selected fleets of the potential short and long-term economic impacts (carried out by the Subgroup on Economic Assessment (SGECA)). STECF is the only source of economic advice DG Fisheries consults on a regular basis. As far as DG Fisheries are concerned, such economic advice is an increasingly important aspect of STECF’s work. STECF does little original scientific work; rather the committee compiles desk studies and reviews the work of others. Whenever original scientific work is carried out, this most often takes place in working groups convened by DG Fisheries to address specific issues [3]: for example, mixed fisheries.

However, regardless of the fact that there is a wide overlap between the experts involved in STECF and its working groups and those involved in the ICES system, the fact remains that these experts—when involved in STECF—are working in context other than ICES. STECF scientists work directly for the Commission. As a consequence, STECF tends to be able to provide advice on issues, and in a manner, which ICES is not—even on issues within its area of expertise. Part of the reason for this is that the same scientists accept different approaches, depending on whether they are working within or outside the ICES system. Within STECF the scientists are free to act more as consultants responding to whatever is required from the customer, DG Fisheries, without having to consider to the extent that ICES does if the requests are reasonable or if answers can be misused.

When the advice from ICES has been through STECF it has gained status in relation to the basic regulation of the CFP, which states that the Commission must take into consideration the opinion of STECF [1]. The regulation has no reference to ICES—or any other external organisation, for that matter. However, the Commission is not obliged to follow the recommendations of STECF and consults others before submitting its proposal to the Council. Two other sources of information are particularly relevant
besides STECF: the Regional Advisory Councils (RACs) and the Advisory Committee on Fisheries and Aquaculture (ACFA).

**Regional Advisory Councils**
The agreement on the legal provision for Regional Advisory Councils (RAC) is one of the most visible results of the recent reform of the CFP in terms of changes in the institutional setup. A recurring critique of the CFP in its more recent years has been its failure to include the knowledge and opinions of, in particular, local and regional stakeholders in the CFP’s management and decision-making process—neither to a sufficient degree nor early enough within that process. This was specifically recognised by the Commission in its Green Paper on reform of the CFP [2]. The provision for RACs responded to this critique as they will function as stakeholder-led advisory forums and will include actors who had, thus far, not had any direct, formal role in the management procedure. It is hoped that including the fishing industry (and other stakeholders) more in the management process will lead to both better decisions, which are less in conflict with reality as experienced by stakeholders, and a higher degree of compliance, due to a feeling of ownership of the agreed rules. In the context of the role of science and scientists, an additional aim of setting up RACs has been to facilitate cooperation and discussion between scientists and fishermen on issues such as data collection and management advice.

In legal terms, the RACs have a purely advisory role *vis-à-vis* the Commission and DG Fisheries. However, DG Fisheries has indicated that the opinions of the new RACs will weigh heavily in the decision-making process in so far as they do not turn out to reflect the lowest common denominator of the fishing industry (interview with high-ranking employee in DG Fisheries, November 2003). How this will work out in practice is one of the main ‘unknowns’ surrounding the RACs. It is also not yet clear in which cases the RACs will be consulted. In the legal texts it states that the RACs should give recommendations to the Commission or national authorities of the concerned Member States on fisheries aspects in the areas they cover. They are authorised to submit recommendations on request from the Commission or national authorities, as well as on their own initiative [1]. The RACs are, in terms of institutional set-up, placed between the Commission and the concerned Member States.

A RAC consists of a General Assembly and an Executive Committee—the latter being of no more than 24 members. In both of these fora, two-thirds of the seats are occupied by the fisheries sector; and the catching subsector of each concerned Member State should have at least one place in the Executive Committee. The last third of the seats are allotted to other interest groups [13]. The fisheries sector is defined as “the catching sub-sector, including shipowners, small-scale fishermen, employed fishermen, producer organisations as well as, amongst others, processors, traders and other market organisations and women’s networks”. The category ‘other interest groups’ includes “amongst others, environmental organisations and groups, aquaculture producers, consumers and recreational or sport fishermen” [13]. Besides these stakeholders, which can act as members, others can be involved in the RACs as experts—namely scientists—or active observers—namely a Commission representative, national administrations’ representatives, a representative of the Advisory Committee for Fisheries and Aquaculture (see next section), or various representatives from third countries. The Executive Committee adopts recommendations,
as far as possible, by consensus. However, if consensus cannot be reached, decisions are taken by a majority of present and voting members. If there are dissenting opinions these shall be recorded in the recommendations [13].

Seven RACs are proposed in the framework-decision, one for each of the following areas/fisheries: the Baltic Sea, the Mediterranean Sea, the North Sea, north-western waters, south-western waters, pelagic stocks, and the high seas/long distance fleet [13]. The chosen areas are indicative of a compromise between a wish to base the management units on biogeographical criteria (large-scale eco-system or fisheries for certain groups of species) and, at the same time, limit their number. RACs for each of these areas/fisheries will be set up on the initiative of stakeholders through the Member States in question [13]. The RAC for the North Sea (NSRAC), which has been operational as from 1 November 2004, was the first to be established [14]. Furthermore, at the time of writing (November 2005) the North-Western Waters RAC and the Pelagic Stocks RAC have been declared operational by the EU.

The RACs’ establishment has contributed to and reinforced a number of discussions, which have evolved around the failings and shortcomings of the CFP. One issue has been the discussion between those advocating a more regional approach to European fisheries management and those who fear that a move in that direction could be a step towards renationalisation of fisheries management. Connected to this is the discussion of whether or not the RACs should have been given real decision-making capabilities. For example, it could be argued that real ownership will only be felt if the recommendations are actually put to use in a more or less unchanged format. A third discussion has evolved around the issue of who can be considered legitimate fisheries stakeholders and what the balance should be between them—a discussion which is, of course, also connected to the other discussions: how different stakeholders conceive and envisage the role of the RACs is closely related to their influence within them.

Advisory Committee on Fisheries and Aquaculture

DG Fisheries will also, in most cases, consult the Advisory Committee on Fisheries and Aquaculture (ACFA), which was set up by the Commission in 1971, in order to be able to take European-level—as opposed to regional in the RACs—stakeholder groups’ opinions into consideration in matters relating to the tasks of the Commission within the area of fisheries [7]. For this purpose, the Committee is authorised to issue opinions and resolutions on various issues and proposals from the Commission [15]. The output of ACFA contributes to the Commission’s knowledge base by providing stakeholder views and information on a broad range of issues. However, ACFA has, according to Lequesne, had relatively little influence on Commission proposals: rather, “[t]he core raison d’être of the Consultative Committee [ACFA] has been an exercise in mutual legitimization” [7].

ACFA, which is organised with four working groups under it, was restructured in 1999 and to a much lesser extent in 2004 to take into account the growing importance of aquaculture in the European fisheries sector [16]. Originally, ACFA’s central plenary committee’s members mainly represented groups with an economic stake in the CFP. However, the 1999 reform brought a wider spectrum of stakeholders into ACFA, which means that the plenary committee now includes representatives of the following
interests: private ship-owners, cooperative ship-owners, employed fishermen, producer
organisations, stock-breeders of fish, mollusc/shellfish stock-breeders, processors, traders,
consumers, environmentalists, and development organisations [15]. ACFA continues,
however, to be numerically dominated by representatives of the fishing industry.

ACFA’s four working groups, which prepare the opinions of ACFA, are: (1) Access to
fisheries resources and management of fishing activities, (2) Aquaculture: fish, shellfish
and molluscs, (3) Markets and Trade Policy and, finally, (4) General questions: economics
and sector analysis. Each working group has a fixed number of members ranging from 15
to 19 allocated from the different groups of stakeholders. The Commission can appoint
additional members according to items on the agenda. In the working groups other
interests/groups besides those represented in the plenary committee are included. These
interests are banks (working groups 3 and 4), auctions and ports (working group 3),
biology (working groups 1 and 2) and economy (all working groups). The representatives
from biology and economics are appointed by the Scientific, Technical and Economic
Committee for Fisheries [15].

8.2.2.2 European Parliament

In many cases the next step of the decision-making process is a hearing of the European
Parliament. This elected body presently consists of 732 parliamentarians from the 25
Member States. However, it is important to note that the vital TAC regulation does not
pass through the European Parliament. This follows from the provisions of the basic
regulation [1]. As this indicates, the formal power of the European Parliament over EU
fisheries legislation is, in general, limited. The consultation procedure, which presently
covers fisheries policy whenever no other provision or specification is made, ascribes the
weakest possible role (besides no role at all) to the Parliament [7].

The consultation procedure requires that the views of the Parliament must be heard
before the Council decides on whether to adopt a proposal on fisheries legislation and in
which form. When the Commission proposes new legislation in the area of fisheries the
Parliament will be asked to propose amendments to it. Most of the work on the Parlia-
ment’s resolutions, which contain the amendments, is done in the standing Committee for
Fisheries. The Committee for Fisheries adopts by simple majority a report as a proposal
for a resolution. The resolution is hereafter dealt with in a full Parliamentary plenary
session, where each proposed amendment has to gather a majority of the votes of those
Members of the European Parliament (MEPs) who are present. If the Commission agrees
with, or at least does not oppose, the Parliament’s amendments, it can change its position
accordingly before negotiations in the Council. The Council is, however, not required
to follow the Parliament’s opinion. From time to time, the Council has actually made a
‘political agreement’ before the Parliament had delivered its opinion. In cases like these,
the Parliament’s resolution is effectively reduced to a ‘rubber stamp’ (interview with
member of the Committee for Fisheries, November 2003).

One way the European Parliament’s Committee for Fisheries and other interested
parliamentarians (in the area of fisheries not that many) acquire knowledge about the
fisheries issues, which it deals with, is through hearings where representatives of industry,
academia, authorities, NGOs etc., are invited. Besides these hearings, MEPs can, of
course, also draw on many of the same sources of information as the Commission, even though they do not have the same access to request opinions on issues of their own choice. Importantly, the Committee for Fisheries is according to Lequesne “the sole EU institution, which uses its reports and hearings to bring alternative expertise to bear on the proposals of DG XIV [DG Fisheries]” and “its deliberate cultivation of expertise as the basis of its efforts to influence the Commission” is striking [7]. NGOs, and to some extent the fishing industry, both of which have traditionally felt deprived of a reasonable representation in relation to EU fisheries management issues, have also utilised the Parliament as a way to gain access to the decision-making system of the EU by lobbying MEPs, who have felt it beneficial to be associated with those interests. However, lobbying efforts have logically been more intense in areas where the Parliament has greater formal influence, such as on environmental policy. It follows from this that the Parliament could potentially become more influential as fisheries issues are increasingly integrated with environmental policy and efforts are made to introduce environmental concerns into the CFP [17].

8.2.2.3 The Council of the European Union

In the Council of the European Union the Member States are each represented by their Ministers responsible for fisheries issues. It is this body which takes the final decision on adopting new legislation. Such decisions include the high-profile setting of TACs, which takes place at the end of December on an annual basis. Fisheries policy, in contrast to many other policy-areas, is dominated by political intervention through ‘regulations’. Regulations are directly binding in the Member States and do not need national legislation to be legally applicable. Fisheries policy is consequently one of the areas where the Council has the most wide-ranging powers, which “virtually amount to direct administration” according to Lequesne [7].

Most fisheries legislation is adopted under qualified majority voting (QMV) where each Member State has a fixed number of votes. The largest Member States have the most votes, but smaller Member States have relatively more votes than the mere size of their populations justifies. In consequence, how often a Member State will be in the ‘pivotal’ position to determine whether an act relating to fisheries is adopted or not, depends on its size and on coalition patterns within the Council. All coalitions are not equally likely to form. The pattern depends on the conflicts of interests in the policy area and the political positions of the governments involved in the negotiations. These two issues can hardly be separated since perceptions of interests will (to some extent) depend on the political positions of governments [18].

Proposals go through a thorough examination at lower levels of the Council before they are adopted (or rejected) by the Ministers. In the case of fisheries, the examination of a proposal starts in the relevant working group of which there are two: the External Fisheries

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5 A qualified majority is reached when the following conditions are achieved: 1) a simple majority of the 25 member states are in favour, 2) the votes they represent constitute at least 323 of the total of 321, and 3) they represent at least 62% of the total population of the EU. See European Union website on the Council: http://europa.eu.int/institutions/council/index_en.htm [Accessed 15 January 2006].
Working Group/Working Party on External Fisheries Policy, dealing with relations with third countries, and the Internal Fisheries Working Group/Working Party on Internal Fisheries Policy, dealing with conservation, markets and structures within the EU. These working groups consist of civil servants representing the Member States. Already at the working group stage, the Commission may consider amending its proposal if there is opposition. When the working groups have completed their work the proposal is passed on upwards in the Council hierarchy to the higher ranking civil servants in the Permanent Representatives Committee (Coreper), which, unlike the lower level working groups, has the authority to decide on questions of a more contentious nature. The Fisheries Ministers will finally deal with the proposals, which cannot be agreed at lower levels. This is always the case for the highly political issue of agreeing on TACs.

Uncertainties in the basis of the scientific advice underlying the Commission’s proposals may be an important element of Council negotiations, since they, to some extent, provide Member States with an opening for questioning, and subsequently revising, proposals, without overtly disputing ‘sound scientific advice’. In this respect, it should be noted that one reason for politicians not to act on scientific advice which will have short or medium term negative socio-economic impacts, is the fact that politicians’ time perspectives often do not stretch much beyond the next election. Consequently, as long as decisive action is judged as likely to be less popular than less significant change, the latter option will be chosen. Where the balance between these options is found is, however, not static. Additionally, there are arguably other explanations for the failure of the Council of Ministers to act upon scientific evidence. For example, efforts by lobby groups might be successful in calling the validity of the scientific advice into question.

8.3 REGIONAL FISHERIES MANAGEMENT ORGANISATIONS

The EU is a contracting party in ten regional fisheries management organisations (RFMOs), which have been set up to monitor and regulate fisheries activities in international waters. The RFMOs can be divided into two groups: those that are primarily defined by the species they deal with and those that are primarily defined by the area they

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7 The number could be argued to be eleven but the International Baltic Sea Fisheries Commission (IBFCM) is currently being dissolved and replaced by bilateral agreements between the EU and Russia. After the accession of Poland and the Baltic States, the EU and Russia are the only relevant members of IBSFC. The ten remaining RFMOs are: North-West Atlantic Fisheries Organisation (NAFO), North-East Atlantic Fisheries Convention (NEAFC), Indian Ocean Tuna Commission (IOTC), North Atlantic Salmon Conservation Organisation (NASCO), Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR), International Commission for the Conservation of Atlantic Tuna (ICCAT), General Fisheries Council for the Mediterranean (GFCM), Western Central Atlantic Fishery Commission (WECAF), Fishery Committee for the Eastern Central Atlantic (CECAF), and South-East Atlantic Fisheries Organisation (SEAFO).
cover. The RFMOs are mandated to decide on management measures within their area of responsibility. This means that the decisions of the organisations are binding on the contracting Member States. The Commission negotiates on behalf of the concerned EU Member States (on issues of conservation of living marine resources) and only if it is not possible to get agreement within the RFMO can a contracting party—for example, the EU—legally decide on unilateral management measures. In some of the RFMOs in the North-East Atlantic, the negotiated decisions are based on ICES advice. Other RFMOs have their own scientific committees, on whose findings, their negotiations and decisions are based upon. However, in general, these scientific committees are heavily dependent on work carried out at the national level and at national expense.

On request, some of the RFMOs provide advice to the Union on fisheries issues within its area of competence. The amount of advice from RFMOs is, however, much smaller than that of ICES. Within the EU system, advice from RFMOs is treated in much the same way as advice received from ICES, which means that it is reviewed by STECF. In the following we shall look at three examples of RFMOs and the science relating to them.

8.3.1 North-West Atlantic Fisheries Organisation

The North-West Atlantic Fisheries Organisation (NAFO) is a geographically defined RFMO with 13 members, which covers most fisheries resources in its area. NAFO’s Fisheries Commission decides on a number of management and control measures based on advice from the Scientific Council of the organisation, which also provides advice directly to the members upon request. The measures adopted by NAFO include, most notably, TACs and quotas for a number of species. The contracting Member States are obliged to provide the necessary information for NAFO to conduct its duties.

To be able to advise on management issues, NAFO’s Scientific Council compiles and maintains statistics for the NAFO area. The Scientific Council also records, publishes and disseminates reports, information and materials pertaining to the fisheries of the area. The Scientific Council consists of assessment scientists from the national fisheries institutes of the Member States. Expenses for these are met by each Member State. NAFO cooperates, furthermore, with ICES on a joint shrimp stock assessment and two ICES/NAFO working groups: ‘Harp and Hooded Seals’ and ‘Reproductive Potential’. The Scientific Council has established four standing committees on various issues [19].

8.3.2 International Commission for the Conservation of Atlantic Tuna

The International Commission for the Conservation of Atlantic Tuna (ICCAT) is one of several tuna commissions worldwide. In the case of ICCAT, the geographical remit relates to the Atlantic Ocean. ICCAT’s decision-making body, the Commission, can adopt different types of management measures, such as TACs, effort control and closed areas. ICCAT’s aim is to keep the concerned stocks at levels where maximum sustainable catches can be achieved. Recommendations and resolutions from ICCAT are drafted on the basis of scientific input from its Standing Committee on Research and Statistics (SCRS).
SCRS is charged with making sure that the Commission has access to up-to-date statistics on fisheries and biological information on stocks. SCRS will also carry out stock assessments. The research activities necessary for SCRS to be able to advise on stocks are generally carried out by scientists in the contracting Member States’ national fisheries institutes or universities. Besides this, ICCAT has a number of scientific research programmes, which are used to help focus, coordinate and complement the national research activities. These programmes, which usually focus on improving biological knowledge or fishery data for particular species, are, in some cases, funded by ICCAT from within their own budget, and, in other cases, funded by contributions from individual signatories or other agencies [20].

8.3.3 General Fisheries Commission for the Mediterranean

The General Fisheries Commission for the Mediterranean (GFCM) is the RFMO that covers most species in the Mediterranean. The structure of the GFCM is, in many respects, similar to that of ICCAT and NAFO. The decision-making body of the GFCM bases its management measures upon scientific input from its Scientific Advisory Committee (SAC). However, the management measures in place in the Mediterranean Sea are in general considered to be weak as, although GFCM has agreed on technical measures—including minimum mesh sizes and gear-type restrictions—it has not been able to agree on TACs and quotas even though they are mandated to do so by the contracting parties.

GFCM’s scientific committee, SAC, has several sub-committees and working groups under it. GFCM also has an aquaculture-equivalent to SAC, the Committee on Aquaculture (CAQ). As in the cases of NAFO and ICCAT, most of the scientific work is conducted at the national fisheries institutes of Member States. Participation in GFCM/SAC activities is, in general, paid for by the Member States as the budget of GFCM is not sufficient to cover the salaries of scientists. The reports of SAC are of interest to the EU, which has important interests in Mediterranean fisheries and SAC advice is utilised in respect to resources within the national fisheries zones within the Mediterranean area [21].

8.4 ELEMENTS OF REFORM

In the face of criticism of the CFP and its perceived management failure—partly ascribed to deficiencies in the advice system—several elements of the knowledge provision system are currently being discussed and to some extent undergoing reform. The reform attempts are related to the institutional setup, the forms of advice, and the communication of advice.

Daw and Gray [4] alongside others (e.g., [2, 22]) criticise fisheries science generally and ICES specifically for failing to involve fishermen more in the advisory process. Increased involvement of fishermen in scientific biological assessment work has been argued to be beneficial in at least two respects. First, scientific assessments would be more credible to the fishermen if they knew that their knowledge was taken into account. This could potentially help to rectify the problem of non-compliance with the present CFP. Second,
the assessment process would be able to deliver more precise results if it had the support
and involvement of the fishermen. The reason for this is that one of the major unknown
inaccuracies of the work stems from misreporting and illegal landings. Such inaccuracies
decrease the trustworthiness of fisheries-dependent data. Fishermen are, furthermore, in
day-to-day contact with the sea and occasionally in a position to be among the first to
notice changes in the abundance of a fish species. It has, however, turned out to be a
very difficult—but not impossible [23]—task to develop meaningful cooperation between
fishermen and scientists. In many ways, these groups come from different worlds and,
even when they do see eye-to-eye, incorporating fishermen’s experience-based knowledge
in management is institutionally challenging [24].

The issue of facilitating cooperation between fishermen and scientists has been high
on the Commission’s agenda [3]. One of the main objectives of the RACs was precisely
to create forums where fishermen and scientists can cooperate and develop a mutual
understanding—something which has hitherto been severely lacking. Consequently, the
memorandum of understanding between ICES and the EU states specifically that ICES
should make every effort to provide scientists to attend RAC-meetings in order to facil-
itate the dialogue between stakeholders [25]. However, no EU funds have been allo-
cated for any (regional) scientific studies that the RACs might wish to initiate. Only
“travel and accommodation expenses of experts attending RAC meetings” are eligible for
funding [13].

The Commission has discussed other possible changes in the setup of the advisory
system. In its Communication on improving the scientific advice, the Commission states
that more resources are needed for the advisory system, as “the overall need for scientific
advice is likely to continue growing” [3]. The system is already stretched beyond its
capacity. In the long term, the Commission suggests two main ways of allocating the
necessary additional resources. One option is to strengthen and reorganise ICES, which
would then have to provide advice on a wider range of issues than today, including advice
for fisheries outside its present area and advice on technical measures. The second option
is to develop and strengthen the internal EU analytical and advisory capacity, possibly
by creating a technical and scientific secretariat to the STECF or to create an institution
modelled after the European Agencies [3], such as the European Environment Agency
(EAA).

Regardless of whether the future institutional setup will turn out to be along the lines
of either one of the two options above, changes in fisheries management approaches
will necessarily change the form of advice needed. Poul Degnbol [26], Chairman of
ACFM from 2003 to 2005, lists several categories of advice, which are at presently
becoming increasingly necessary at the expense of the traditional single stock catch
projections:

- Advice relating to the implementation of the ecosystems approach. This entails inter
  alia establishing ecological quality criteria.
- Advice, which relates to long-term management. An example could be advice relating
to the multi-annual recovery and management plans of the EU.
• **Advice relating to effort (input) regulation** rather than TAC (output) regulation. This is also closely related to the adoption of EU multi-annual management or recovery plans, which in most cases involve effort regulation.

• **Advice relating to fisheries rather than single stocks.** Much of EU fishing is not suited for management by single species TACs because most demersal fisheries are mixed. Advice on mixed fisheries effects is therefore important.

The above mentioned issues are touched upon in the present memorandum of understanding between the EU and ICES. The memorandum states that ICES should move towards multi-annual advice in line with the provisions for multi-annual recovery and management plans as set out in the basic regulation of the CFP [25]. Fast-track advice, which is not mentioned above, cannot really be considered a new form of advice, rather a speedier way of providing advice. Access to fast-track advice is a high priority in the Commission. This type of advice is considered of vital importance when stock levels are low—as they presently are—and conservation measures are urgently required [3].

Finally, how scientific advice is communicated has also been under discussion and will probably undergo changes. Scientists have been accused of not being sufficiently transparent in their development and delivery of advice. This lack of transparency, which has been noted in several places [3, 4], undermines the credibility of the advice among stakeholders, who may very well feel that uncertainties and other important factors are kept hidden from them. ICES is, as a consequence, currently working to make its procedures and advice more transparent.

### 8.5 CONCLUSION

The system that provides scientific and technical advice to the decision-making process of the CFP is highly complex and dynamic. The system is currently undergoing reform and will most likely continue to evolve. Within the EU context, which has been the focus of this chapter, the reform efforts are elements of an overall reform of the CFP, which has been brought about by the critical state of many important fish stocks, as well as by the problematic situation of the industry. However, the reform of the advice system is also driven by global developments relating to changes in the perception of how to manage fisheries efficiently—for example, the change towards an eco-system approach to fisheries. It is difficult to predict the outcome of the present reform but it is clear that there will not be a lesser need for qualified scientific advice. Rather, new demands on advice relating to the eco-system approach might very well put more demands on an already overstretched system.

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REFERENCES


Chapter 9

Participation

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9.1 DECISION MAKING THEORY

Since the Second World War, public policies have been closely aligned with the development of the welfare state and the need to address the causes and effects of economic cycles, as well as those events that, in economic and social science circles, are classed under the generic heading of market failures. One of the basic tasks of public decision-makers should be to provide a solution to such questions and guarantee the stability of the economy and society as a whole.

Public policy decision-making, viewed from the economic perspective, revolves around two key features. Policies are grounded in an approach known as ends-means rationality. From this approach, in the early 1950s, there gradually unfolded a theory that remained predominant for over three decades—the public intervention theory. This methodology first identifies the ends or objectives to be pursued and then selects the best means of attaining them, in two separate steps. The responsibility for developing public policies belongs to the regulator, which has always been the institution legally charged with the task. In some cases it has fallen to the state, in others to federal or regional administrations.

To establish their ends or goals, representative democracies have customarily resorted to a political process through which collective preferences are determined by way of a suitable electoral procedure, in which individuals are able to express their preferences. The results reveal the collective preferences from which the social welfare function can then be derived. The second part of the process is to maximise social welfare using the technical instrument best suited for the purpose, to the exclusion of any other criteria. The outcome of this procedure is a welfare-optimising public policy that enables the public authorities to decide on the optimum allocation of available resources. This public intervention theory placed the regulator in the role of mere specialist intermediary between collective wills and their technical and scientific transformation into practical public policies.
From a global perspective, resources are allocated by means of a mathematical optimising technique or, in any event, a formal technical mechanism, in which the regulator solves the problem by simply applying the best available calculating techniques. Whatever the difficulties involved in the calculation process, or in the design and implementation of the political process used to determine collective preferences, the role of the regulator, in this scenario, is restricted to the mentioned function.

Public action was considered absolutely neutral and technocratic. Thus, its legitimacy was always based on the fact that it was the means by which optimal policies could be developed, using totally objective criteria, in which neutrality had to be guaranteed by the impartiality of the procedures used in the process. The performance of policy-makers was evaluated only in terms of their capacity to efficiently attain the goals determined through the electoral process. Preferences were the prerogative of individuals and the community; once their wills had been expressed through that process, the regulator, as a specific agent, should be free of any preferences. In so far as the welfare theory largely solved the basic problems of the optimisation programme, it was along these parameters that it pursued a long-lasting discourse [1].

From the arguments that have been put forward about regulation and regulators, it is easy to see that role of scientists, experts and knowledge in this model had to be to provide the scientific and technical analyses needed to devise an optimising programme technically suited to the preferences expressed by the collective will. From their neutral position, their role as a part of this process should also include proposing designs for optimal policies, which policymakers are then obliged carry out, always accepting they are not allowed to interfere in the public decision-making process.

Historically, economics and decision theory have been firmly split between two subfields or branches: positive and normative. The positive branch takes the main weight of providing the basic analyses to explain phenomena and supply the concepts and models needed to understand facts. The approach of the positive branch of science has been to maintain a basically passive role with respect to facts. As such, it has not involved itself deeply by intervening with its models in the analysis of the concrete realities that surround us.

The normative branch, meanwhile, has essentially taken up the task of applying the methods and models proposed by the positive branch to the melting pot of practical problems that are raised. As such it has taken responsibility for the applied aspects of resource science and management, preparing the best means to achieve the goals demanded by the policymakers, while dealing with the particular complications of specific realities. This specialisation in the practice of the two branches showed up quite sharply during the period mentioned, and served as a basic cognitive model. This conception of public policies provided the basis for the regulative procedures referred to in the literature as top-down models. Indeed, as already indicated, in this model the regulator acts as an all-powerful, providential agent who assumes responsibility for the planning and implementing the policies.

From the simplified explanation given above, it is easy to foresee the kind of criticism this approach was liable to receive. It is a frame of reference within which at least two key agents in the public policy-making process were denied any opportunity to express their preferences and, therefore, had to feature as neutral agents in policymaking.
While complying with what public policies have calculated to be the plan needed to maximise social welfare, the different agents in society see their position become uncertain when some of the agents discover that some policies fail to satisfy their interests. When this happens, findings suggest that those agents who see their aspirations unfulfilled react strategically, and neglect the plans assigned to them by regulators. In fact, they tend to place their own interests above or on the same level as the theoretical common goal proposed by the regulator.

Criticism from different quarters demonstrated that the ‘conventional model’, based on public intervention theory, suffers from an absence of any analysis of agents’ interests. As a result, theorists have gradually introduced the idea of the importance of group action by agents sharing common objectives when it comes to public policy-making. At the same time, however, agents were considered as a set of indeterminate individuals with neutral interests or as a set of individual interests interacting with a global outcome, which, in equilibrium, neutralised their mutual influence over public policies.

Acknowledgement of the active presence of individuals, social groups or institutions with their own objectives, separate from the collective welfare objective, as defined by the regulator, represents a break with the idea that each should passively comply with the optimising programme devised by the policymaker. From this context, there arose the need to analyse the behaviour of agents and groups, all defending their own objectives and the impact they would have on the decision-making processes. This analysis, which enabled the recognition of the importance of different social groups and communities and of their influence over policymakers and public decisions, was extended to include the regulators themselves who had, until then, been seen as the essence of neutrality.

The natural extension of the notion of self-interest groups to include apparently super-neutral agents highlighted the need to develop a more global theory of the participation of stakeholders involved in public policies, whether these be general or sectoral. This is particularly important in the European context, and even more so in sectors where decision-making has been centralised in the European institutions, as is the case in the primary sector.

9.1.1 The incorporation of participation in modelling

Successive criticisms, and the evidence that has been found to support it, have generated new proposals to incorporate participation in the models used to explain the public policy-making process. It is clear that an inclusive approach has prevailed among the social agents, who have diverse interests participating in the various stages of decision-making ([2] pp. 59–77).

In some cases, and in some economic and social sectors in particular, the interplay of audiences and participation patterns has been organised on an institutional basis and a stable institutional organisation has been created. In other cases, developments have been informal, with varying degrees of transparency. Regulators (from, for example, state and regional governments and parliaments), who are formally charged with the responsibility of adopting the decision, work alongside them. These agents also have their own interests and act as key players in the game of pressure.
Even the state administrative bureaucracy often ends up taking part. In principle, the administration might be thought to play the dual roles of expert advisor on technical aspects of decision-making and guarantor of the implementation of policy decisions. However, despite apparently having no stake in the decision, the state administration very often ceases to act as a non-stakeholder and, instead, uses pressure to protect its own interests. It has also been seen to display a tendency towards competitive bias in budget distributions.

Contemporary civil society, moreover, tends to be constituted by a proliferation of groups organised around different objectives. These are, very often, non-profit seeking organisations with interests linked to the various types of deficit that exist in modern society. All are engaged in pursuing specific objectives and all use pressure to sway public policy decision-making, sometimes to the point of mobilising their members and supporters to wage a campaign of resistance [3].

In addition to these groups, social organisations, with a long tradition of being interested in the structures of economic and social activity, are also involved in the policy process. These include unions, employers organisations, and various types of professional associations. Historically, the research community has been attracted by social organisations from the primary sector—in particular farmers’ and fishermen’s associations—and in professional bodies within the public administration, who all take an active part in public policy design.

It is also important to bear in mind the receptive attitude that decision-makers have developed towards these groups. They have no doubt learned both that the various agents can serve as providers of useful and interesting information, but also that they possess an expertise that can be properly channelled only by involving them in a participative model. Thus, organisations drawn from civil society have gained increasing recognition as an instrument by which the improvement of the decision-making process can be achieved and as an important legitimating factor for decisions affecting social and political life. As such, they have been included in the deliberation process. Thus, it is of interest to consider their importance and establish how they can be made to fit into the public policy-making process. This places participation at the forefront of the analyses and concerns of both theoreticians and practitioners.

9.2 EUROPEAN INSTITUTIONS, PARTICIPATION AND KNOWLEDGE PROVISION

The dynamics of the development of the EU have resulted in a multi-tiered institutional model in which the historical institutional structures of the Member States co-exist and interact with those that are being created and developed within the Union. This institutional framework has given rise to a polycentric, multi-tiered decision-making system, which differs from the traditional state system, without actually creating a new state or new government in the traditional sense of such terms. While the institutional and administrative structures of the Member States, with their corresponding sub-tiers at regional and local level, remain valid, there now also exists the European level, with its own institutions and its own way of dealing with the relationships involved. Thus, the
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decision-making model used in the EU has given rise to a complex pattern of co-ordination between the institutions involved and between the various vertical and horizontal tiers of government. These changes have no doubt had a profound effect on the functions fulfilled by the public policy-making centres at the state or regional level, which, with varying degrees of difficulty, have managed to adapt to the new demands [4].

Thus, in the space of a few decades, a continental-scale Union is being created. This has seriously challenged our capacity to develop and implement public policies that reach beyond the boundaries of Member States. Through consensus-building techniques in the complex structure that has been set up, it has been possible to construct the Union as we now know it. It is precisely the way it is put together that has turned it into one of the hubs of the globalisation process currently taking place world-wide.

Nevertheless, some of the problems, incoherence and uncertainty, which agents have to face, arise precisely out of the difficulties involved in putting into practice the policies generated by such a complex machinery. Its complexity is reflected in the problems that have to be addressed at the various stages of policy-making, starting with their scientific and technical conception, through the preparatory stage, the administrative work, and so on. The inherent characteristics of the methodology employed are such that, as a result of the difficulties involved in dealing effectively with such complexity, legitimacy issues are raised by the public, who are often dissatisfied with a model that strikes them as distant and complicated. Part of the knowledge base should be used to highlight the difficulties involved in conducting a participative process within such an environment, identify possible sources of failure and the options made available through its incorporation.

Given that the EU is not grounded on the traditional mechanisms of democracy in operation in the Member States, it cannot rely heavily on political representation for its legitimacy. In this respect, much of its legitimacy may stem from the effectiveness of the public policies it develops, something which, in many sectors, is not easy to achieve. There has been notable difficulty in achieving positive outcomes from the public policies affecting the fishing sector, for example.

Fishing and agriculture were the first sectors in which the responsibility for public policies was transferred from the Member States to the Union: thus, they have been regulated at Union level from its earliest stages. The first rulings that were eventually to make up the CFP began to be adopted in 1970. The distance created by the transfer of power from the Member States to the European decision-making centre reinforced its top-down dynamics and drew even more attention to the effects of the hierarchical regulation model. In a large number of European countries, the various organised stakeholder groups within the fishing industry, basically those of the harvesting sector, began to barricade themselves ever more tightly into their professional and sectoral organisations at the state level. These organisations have not managed to develop a routine of interrelation and open intervention with the European centres for the creation and implementation of fishing policies: particularly European Commission. As a result, the industry has never had representatives who could influence the policies created by the Commission. Nor have European decision-makers ever succeeded in gearing their policies to match the interests of the key agents within the sector.

In the fishing industry, participation at European level has been very superficial. Any exchanges that have taken place have been between privileged spokesmen, such
as Ministers of fisheries and fishing administrations in individual Member States. And so, the fishing industries of the individual Member States have turned to the Council of Ministers in their search for an opportunity for talks at the European level. While doing so, the industry has conducted a strategy of putting on the pressure once decisions have been taken, but remaining absent from the arenas in which they are discussed and developed.

This process has revealed the presence of a two-fold crisis. On the one hand, neither state nor European level regulation models have, as yet, succeeded in creating the proper channels to integrate the fishing industry’s stakeholders and to link the different levels and centres of decision-making in a coherent manner. As a result of this failure, for a long time, the only policies that were introduced were vertical policies, the legitimacy of which few of their targets recognised. Fishing industry firms, meanwhile, also lacked a sectoral structure on a European scale that would allow them to consolidate their interests at that level of action.

All these institutional factors that enabled the persistence of the top-down structure of fisheries management in Europe give rise to the idea that the crisis involved in finding the right model of governance could be held largely to blame for the crisis in stocks which European fisheries have suffered in recent decades. At the same time, the industry’s stakeholders have stressed the need to introduce new structures that do not emanate from governments or public administrations; and for themselves, and other stakeholders and civil society organisations affected by the polices, to participate in fishing policy making processes.

In the European setting, a strategy known as governance has been devised in an attempt to ensure a new model of governability for economic sectors and society as a whole. One of the key features of this governance model is that economic and social agents are assigned an input, enabling them to participate to different degrees in the creation and implementation of public policies. These governance models are characterised by the fact that the rules of play, which are different from those that have prevailed traditionally, are shaping new ways for the representation of both general and private interests. In the area of knowledge-base provision, the various stakeholders are able through their participation to bring elements to the participative process that would otherwise not be introduced into collaborative knowledge creation processes.

9.3 DIFFERENT APPROACHES TO THE PROBLEM: MODES OF CONSTRUCTING A COLLABORATIVE KNOWLEDGE BASE

The quality of outcomes from scientific models, and of related proposals for the regulation of the fishing industry, depend, in part, on the quantity and quality of the available data and models to make the knowledge system work. It is, therefore, essential to ensure that industry actors collaborate in the process. However, their degree of involvement may differ [5]. The different levels of participation are presented below in ascending order of the degree and complexity of commitment. These models of collaboration are cumulative in that each one incorporates the basic perspective of the earlier ones. The scheme is based on Wilson [6] and was developed from collaborative research, though we use it
here for participatory approaches to creating the management knowledge base. This is laid out in detail in Chapter 4. In summary the four models of collaboration are.

*The Deference Model:* Scientists are the experts and the best way to get an accurate picture of nature is to rely on their professional judgement. *Experience-based versus Research-based Knowledge:* Scientists and fishers see the world differently because of differences in training, experience and culture. Science can be improved by listening to what fishers have to say about the resource. *Competing Constructions:* Scientists and fishers collaborate within interest groups to create pictures of the resource that reflect particular concerns. *Community science:* Collaborative fisheries science conducted in the context of cooperative management.

### 9.3.1 Deference model

The notion of deference has been studied in various contexts (such as, organisational, semantic, cognitive) and some of its connotations are common to all, while others are found in one context and not in others [7]. In the case in hand, this notion will be considered in the specific sense proposed by Wilson [6]. Deference, as it is understood in the present context, therefore, refers to a specific attitude on the part of industry actors, in particular those of the harvesting sector. It is the attitude that actors display when they subordinate their own interpretation of activity in the fishing industry to that of third parties. The observations made by these actors over years of activity in the industry, their conjectures and systematisation of the knowledge acquired from experience, and their stock of empirical learning, are all subordinated to scientific knowledge. Thus, they take on a subsidiary role in regulatory policy-making, adopting an attitude of deference, and allowing scientific knowledge to prevail over all other forms. They accept that scientists and scientific data play an essential role in order to interpret reality and make the models operative. This is what makes them comply with the demands of scientists engaged in assessing stock dynamics or making TAC recommendations.

The industry actors in this case take the view that scientists are cognitively better equipped than they are to generate proposals and to suggest possible solutions. Scientists’ analyses are understood to have been conceived from the universal body of scientific knowledge and to be better adapted to regulatory demands than those based on empirical knowledge, which is local or non-universal. It is thus accepted by fisheries actors that the effectiveness of scientific methods and tools has been proven and is beyond doubt. Thus, those who are properly instructed and trained in the use of such a tool are able to achieve a fuller understanding of the facts, enabling them to develop general proposals with the accredited potential to transform reality. As a result of this acknowledgement, fisheries actors defer to the knowledge tools in the hands of scientists. While the industry actors may not understand the specific procedures and may be unable to apply the techniques involved, they trust the scientists to solve problems of both a specific and a general nature.

Another aspect of deference is that, for collaboration to work, scientists and the industry—often with encouragement from the public sector—need to agree on a plan of action, which requires a framework of sustained collaboration. This has been achieved in many instances and many countries. Thus, within the scheme of this collaborative
knowledge mode, industry members assume the task of compiling information drawn out of their activity, in order to satisfy the requests of scientists.

This, in synthesis, is the acknowledgement of science as a tool that fisheries actors, without directly understanding it, nevertheless rely on. Such reliance is possible because society has accepted that if it follows the rules of play laid down by scientists, assuming them to have been correctly and scrupulously formulated, this tool can lead them to great achievements. In summary, industry actors have no direct understanding of the techniques but nevertheless defer to them.

9.3.1.1 Some examples of deference collaboration

A practical example of this model of collaboration in knowledge production is the process of tagging. In a work about tagging of bluefin tuna (*thunnus thynnus*) Cort [8; 36–49] relates bluefin-tagging experiences in Europe during 1911 and 1991 and he explains his own campaigns between 1976 and 1991 with 5763 units tagged. He explains the excellent collaboration he could obtain from several fishing vessels using a live bait method in the Cantabrian Sea. This is a practice used extensively with all the species of commercial interest. It constitutes an archetype of deferential collaboration.

In the last ten years, the collapse of the codfish has lightest off alarm bells and numerous campaigns of cod-tagging have been undertaken. In Canada and the USA, the Regional Northeast Cod-tagging programme was undertaken during 2003, and, recently, another programme of tagging has been undertaken in the Baltic and North Seas. Certainly, contemporary tagging programmes use very sophisticated mechanisms: able to measure the depth, the salinity, the temperature of the water in which the fish swims several times each hour. Nevertheless, the collaboration procedure remains essentially the same as that employed in traditional tagging programmes. To relate this to industrial fisheries, the presence of scientists on board fishing vessels is very common and there have been a variety of experimental and research campaigns carried out on a collaborative basis. For example, an important campaign was undertaken within the European Union Bigeye Programme to collect by-catch information on the European tuna purse-seine fleet. Between June 1997 and May 1999, a total of 62 observers’ trips were conducted as a part of this project and it is considered the largest observer programme ever carried out in the European tuna purse-seine fishery [9].

9.3.2 Experience-based knowledge (EBK)

9.3.2.1 The rationale for such a term

The term “Experience-Based-Knowledge” comes from the generalisation of a succession of more restrictive concepts, each of which represents a different aspect of the same idea. Thus, Traditional Ecological Knowledge, Indigenous Knowledge, Native Knowledge, Fisher Knowledge, etc., each highlight different aspects of knowledge creation and transmission. Some place an emphasis on the experience held by a community through the activity of past generations. Others focus on the nature of the community in which the experience has accumulated, its main cultural features and the specific values that
distinguish it from other fishing communities, who have in turn acquired their own knowledge over several generations. Others aim to illustrate the importance of the experience acquired, not by the community, but by individual fishermen throughout their working life. Here, issues such as the specific characteristics of women in fisheries also come into play. Nevertheless, the point of convergence of all these concepts is that experience is the basic element that they contribute to knowledge creation.

9.3.2.2 The difference between deference and EBK

Experience-Based-Knowledge (EBK) is a further step in the evolution towards an understanding of the collaboration of the fishing industry with the scientific community in the production and use of resource users’ knowledge.

In the deference model, co-operation came from the recognition of the capacity of science to assess the facts. It was accepted that responsibility for the preparation of regulatory plans and methods should be grounded in scientific method. Collaboration with the needs of science was therefore offered, under the assumption that public policies that had been developed within scientific parameters would be both influential and effective. Fishermen’s role in deference knowledge production was limited to providing technical input and the data needed to develop it properly. The EBK-type collaborative strategy, however, involves the integration of the knowledge possessed by fishermen and other stakeholders into the scientific model. The long-lived relationship of such actors with the environment in which they work and live provides them with close, detailed knowledge of the key variables of various aspects of the industry, which scientists do not usually possess. This type of knowledge is frequently related to ecological knowledge: the conditions in which fishermen work and the way in which they coexist with the different species. Thus, as well as providing the data needed to make the chosen models work, the aim of EBK is also to broaden the scope of existing models. Of course, the scope of EBK does not include proposing a model to replace the scientific one. It was never intended to do so. It was neither conceived nor authorised for such a purpose.

9.3.2.3 The need for new contributions to the traditional scientific model

Though this type of collaboration can be extended to include stakeholders from different sectors of the industry, its clearest and most successful manifestations have been seen in the harvesting sector [10]. A basic issue to be borne in mind with respect to stock assessment and stock dynamics is that they are performed by indirect means, using long-term catch records. For this reason, scientists use catch levels as a direct input in their models, but there are many other related data that fishermen could provide but which, normally, no one demands. Traditionally, it has not been common practice to take into account the knowledge of the agents who make those catches, or to ask for information regarding, for example, the strategies used or the species involved.

Some recent failures in the regulation of major commercial fisheries have brought to light questions about the nature of fishers’ knowledge. Meanwhile, serious threats hang over a large number of commercially important fisheries. Both of these situations reveal the need for further research into current methods and knowledge sources, in order to address regulatory issues with new management tools. Failure to predict the dynamics of
some stocks and the difficulties involved in obtaining accurate estimates of the situation of others have led numerous analysts to change their focus. They are now beginning to question the uncertainty and ignorance that prevails in some areas of our knowledge of fisheries and the congruity of key aspects of our public policies. This is one of the reasons for importance being attached to the need to incorporate other forms of knowledge into the traditional scientific model. It has also provided one of the main motives for the switch of attention towards the knowledge of the actors involved in the use and exploitation of fishing resources.

One explanation for the historical persistence of this lack of awareness is that scientists have traditionally considered such knowledge to be superfluous to the requirements of their scientific method. It is true that the two approaches employ very different building procedures and the disparity of their formats has for a long time led to the idea that there is only a very loose link between them.

Indeed the fisherman’s goal is ultimately tied to the success of his economic project, which is why his familiarity with environmental specifics and knowledge of the species he fishes are so important to him. The keys to a good catch have a lot to do with the fishermen’s knowledge of the habitat of the fishing grounds, the choice of species for each time of year, the market price of each species, and the areas where they are most likely to be caught. The fisherman weighs up his chances of catching each species or combination of species and makes the choices he considers will bring him the most benefit, taking account of his restrictions. This is what makes the difference between a working team successfully conducting a project in an efficient firm and another failing in the attempt. Scientific knowledge, meanwhile, concerns itself more with the laws and patterns that can be drawn from the different disciplines that study fishing activities and species. When it comes to stock dynamics, scientists are concerned, above all, with the evolutionary patterns of fish populations, an issue in which the geographical area in which the catches are made is relatively unimportant; or at least much less important than it is to the fishermen.

Since the fisherman’s knowledge is based on his perceptions of the environment in which each species lives and develops, his main concern is with getting to know specific details of the spatial and temporal environment of each species. In fact, fishermen deal with fishing resources on a much smaller scale than the regulator or scientist because their relationship with those resources takes place in the playing field. Fishers’ catch strategies are designed on the basis of the knowledge acquired through a long-run, cumulative interaction with the different species and the different catch strategies employed. This places them in a privileged position to learn about the characteristics most pertinent to their work, such as the places where each species can be found in greater or lesser abundance, its living habits, the most and least favourable ocean parameters to locate and catch each species, and so on. This provides them with what are, basically, local and qualitative assessments of the nature and activity of the species. There is a basic set of tools—including radar, echo-sounders and decca—which is used to measure specifically these variables, and the skipper keeps a log of the location of the fishing grounds and catch sizes over the different seasons. In this respect, the mechanisms that give the resource user his knowledge serve an essentially different purpose from those used by the scientist, and the nature of the research made by each depends on their priority goals. In EBK,
quantitative accuracy does not in principle need to be very high, because that is not its chief purpose.

Fishermen’s information and knowledge builds up through empirical testing over relatively long periods of time. It develops during their daily activity in the medium where the activity takes place. Their professional activity and their increasing knowledge-base on catches and resources form a complex unit in which individual and collective identities are developed. The scientist, meanwhile, understands knowledge production as coming from a universe of generalisation. Scientists perform in a scenario in which they play the role of observers, rather than actors; they are more concerned with general rules and less with the specific behaviour of a stock in a particular place at a particular time or the details of the habit of a certain species. Their main concern is to understand the mechanism behind the resource behaviour, from the scientific perspective of their particular discipline. Their aim is to obtain both qualitative and quantitative assessments [11].

Research goals also differ across the two perspectives. For the fisherman, the learning and transmission of knowledge, and the way it is used with the species that he fishes, is a way of introducing cognitive certainty and security into a natural and social environment that is risky and competitive. Thus, the ideas and expectations, forged through the information exchange, help to form a mental positioning system or cognitive map to guide and control his performance in an uncertain and unstable environment. In this way, knowledge and information bring some regularity and order to the uncertainty surrounding the fisherman’s adventure. In this respect, it can be said that the flow and exchange of information generate predictable and regular behaviour in the fisherman. They replace randomness and error with predictability or redundancy.

The fundamental virtue of EBK is that it can provide qualitative—and even quantitative—data for certain key indices and components. This contrasts with the difficulties of traditional regulation in aspects relating to the heterogeneity—in terms of feeding, breeding, habitats of repose, maximum mobility—of the medium in which stocks develop. This heterogeneity was not taken into account in the traditional scientific model, where such differences were usually considered irrelevant to the central issue. Despite this, their importance was illustrated by Wilson et al. [12]. Thus, in the normal course of events, the conversion of experience-based knowledge into scientific knowledge requires proper translation, so that the two logics can converge into operational resource management models [13].

A major boost to the legitimacy of the integration of EBK in knowledge production came from the recognition of this approach by the United Nations. In 2002, the UN proposal, Agenda 21 for Sustainable Development, pointed out the need to incorporate EBK into the core of scientific knowledge in order to further understanding of natural systems. The proposal also drew attention to the limited capacity of the existing scientific knowledge schemes of the time to evaluate natural resources. In consequence, it announced that further progress should be made in that direction, indicating the need for international assistance in the promotion of sustainable development. Since information of the EBK type was considered essential for such a task, its potential as a genuine source of knowledge became recognised at international level, and, as such, it was merged into some conventional scientific programmes.
9.3.2.4 Examples of EBK

In the example to illustrate the deference model, Cort [8] indicates another interesting variant of collaboration that fishermen offered him during the tagging campaigns. Fishers were suppliers of empirical knowledge and providers of conjecture and hypotheses related to several aspects of tuna behaviour, such as their preferred oceanographic and environmental conditions. For Cort, the information derived from tagging and the knowledge provided by fishers and by fishermen’s professional organisations was of valuable importance to his PhD. In this work, he identified the cognitive structures of fishermen, which have proven their validity to make predictions on different aspects of the fishing system and to use their marine knowledge to secure the survival of their fishing operations.

A somewhat different experience was found on the Galician coast by Freire and García-Allut [13] From information provided by the artisanal Galician fisherman, Freire and García-Allut made an in depth study of the microhabitats of key species. Using information from fishers as a starting off point, they made an X-ray like map of the seabed and established what species were living there and what their habitats were. In addition, they could explain fishers’ knowledge base and their socio-economic adaptations and fishing strategies in those microhabitats. Freire and García-Allut explain how such expertise, which has accumulated in the community, makes it possible for them to live from the heritage that passed on that knowledge. After this experience, they developed a software package so they were able to apply that methodology to other case studies.

Other interesting cases of EBK collaboration were the participation of the fishermen and their professional associations in the election of the most suitable places to install artificial reefs in the Cantabrian Sea during the 1990s. While, in the Mediterranean, around the island of Menorca, and, in the Atlantic, around the island of Hierro, the election of Marine Protected Areas (MPAs) was conducted in collaboration with fishermen and their professional associations [10].

In some social science disciplines, it is very common to make use of extremely varied sources. In some such studies, the proportion of oral testimonies, in relation to other sources, and the importance attributed to them, is greater [11]. There are numerous case studies of this nature have emerged from the fields of ethnography, anthropology and sociology. Another study regarding fishers’ behaviour when organising work teams in the Cantabrian artisanal fleet is based upon significant oral research [14].

9.3.3 Competing constructions

9.3.3.1 Fishing economics as a normative science

The principal objective of analysis conducted by the various scientific disciplines in fisheries over the last fifty years has been to investigate the causes of over-fishing crises. During this time, the different areas of fisheries science have been engaged in trying to provide applied regulatory methods and public policy proposals to deal with the problem of over-fishing. Among all these applications of science to the regulation issue, however, one model has proved extraordinarily successful. This is the bio-economic model, which,
throughout the whole of this period, has displayed a clear normative perspective. The basic aim of this model has been directed specifically at regulation problems [15].

One of the most important points made in the conclusions drawn from the bio-economic model is that over-harvested fisheries (known as ‘free access fisheries’) result in an automatic resource revenue loss and, that, in such open access fisheries, the productive capacity of many fisheries, sooner or later, exceeds their reproductive capacity. A large number of public policy instruments have been devised to deal with the over-fishing problem and to correct this revenue loss.

During this period, the scientific approach towards fishery management problems has become increasingly inter-disciplinary. Fishery management was, first, approached as a biological issue; and the economic perspective was added later. Used in conjunction, these two approaches have given rise to the bio-economic model, upon which public policy has been largely based in recent years. This bio-economic model has gained outstanding relevance, moreover, because it gave birth to a specific discipline within fishery regulation. This model has enabled us to link biological concepts, such as the natural mortality rate, fishing mortality rate, birth and growth rates of species, expressed in quantitative terms as weights, with concepts such as revenue from catches, fishing effort costs and economic returns from the resource, expressed in monetary terms.

The mathematical expression of this model and its subsequent extensions have enabled advances in the two disciplines to be linked together in a common formal language. This has provided a good means by which to capture the sense of problems and to meet demands raised by regulators, while developing a pool of knowledge from which a specific framework has evolved. The advantage of this procedure is that it has provided a common quantitative and qualitative language for public policy. For over three decades, the bio-economic model has been the dominant paradigm in the realm of fishery regulation. Since its first conception, and throughout its subsequent development, it has been permanently available for the regulation and planning of fisheries. Thus, its orientation has been manifestly normative and applied; and it has been the regulator’s instrument of choice. Thanks to its availability and capacity to provide solutions based on precise quantitative and qualitative proposals, its legitimacy has been fully recognised throughout the whole of the fishing industry. The recommendations derived from this perspective usually provide a clear and relatively concise mandate, which allows the regulator to make public policy proposals and to reach clear decisions. In this sense, it is also highly appreciated by the regulator, whose decisions tend to be aimed at limiting fishing effort and sustaining resource revenue.

It is worth stressing the interdisciplinary approach and the symbiosis of disciplines involved in the bio-economic model. The importance of this for the scientific context and the regulatory scene is, without doubt, comparable, disciplinary differences notwithstanding, to that of biochemistry, for example. It has additional value, moreover, in that it serves as an example for more extended symbioses currently being tested in other disciplinary areas of fisheries science. Much energy is being devoted to developing this interdisciplinary approach and the isomorphism of the bio-economic model is being extended to new fields and disciplines and further developments within the original bio-economic model.
Another aspect to the fisheries management problem, meanwhile, is that, due to its initial characteristics, the bio-economic model has resulted in a top-down centralised way of managing fisheries, similar to that critically analysed in the first section of this study. They reveal the same methodology in the way that several important agents were lacking in the decision-making process. The successive incorporation of participants with their competing interests and their competing constructions has developed a more inclusive approach to policy-making.

The customary regulation models of the past—which were designed for barely industrialised fisheries—often dated from far back in history and were based on decentralised resource allocation mechanisms. Such models functioned as self-management regimes and the principal stakeholders in the sector were usually involved in the running of them. These regulatory models began to break down, however, as the fisheries became increasingly industrialised and pressure on resources led to general overexploitation [16]. At the same time, bio-economic models gradually replaced the traditional type, and, after a long period during which this last methodology has sustained the centralised type of regulation, new bottom-up competing perspectives have emerged in fisheries policy making.

### 9.3.3.2 Participation with competing constructions

When the views of the interested parties are introduced into the problem-solving process, significant changes take place in the scientific formulation of the issues. In addition to the issues raised by scientists with respect to reality and the questions posed by the regulators, it became necessary, in the new set-up, to address those brought up by individuals, interest groups, or others affected by the decisions. In other words, attention had to be given to the issues perceived from the stakeholder perspective. In this new scenario, participation became even more important, both in the decision-making process and in the preparation of the scientific research agenda by stakeholders. In this model, the boundaries between scientific output and management are not as strict as in the bio-economic model and there is a growing need to construct a framework in which the different scientific disciplines can collaborate in seeking the answers to the questions that arise.

Thus, the participation of stakeholders has led to a formula in which the means for generating regulatory proposals have become enormously complex and the classic biology-economics twofold method has been invaded by new disciplines. At the same time, this development has led to a significant opening up of the regulatory framework where, formerly, only the harvesting sector, professional fishermen and the state regulatory bodies were privileged to speak. A great variety of stakeholders have joined in the public policy-making process, which now includes processors, wholesalers, recreational fishers, and representatives of civil society such as environmentalist groups. All of these have incorporated their own interests and values into the collaborative agreement on public policy design, with the result that the scientist’s role is no longer reduced to preparing what Pellizzoni [17] has coined as “the best argument”. Very often, participants take up conflicting, as opposed to simply competitive, positions. It is rare for stakeholders to put forward their own assessments of certain strategic variables to address the regulatory problem, with proposals to collaborate in stock assessment for example, or even possibly in setting TAC levels and deciding on quotas.
Scientists will be required to devise alternative scenarios to allow for the diversity of positions that now converge in the public policy-making process. It will, therefore, be necessary to establish a basis of collaboration for all the stakeholders, both in knowledge production and in certain stages of the decision-making process.

Through the range of collaboration models, from the deference model, through EBK and particularly in the competing construction model, participation has become increasingly inclusive. It now does not only feature demands of those directly involved in the industry; it is being extended to include all those affected by decisions, who, in turn, are contributing their own demands. As a result, we are now beginning to see demands from consumers, environmentalists, and defenders of gender equality, for example, with the result that, as participation is becoming more extensive and inclusive, the public policy-making process is adopting a bottom-up approach that will inevitably make it more complex. In this respect, the competing construction model includes and envelops both the deference model and the EBK model, while also inviting the collaboration of all industry stakeholders in the creation of new knowledge.

The significance of this change in the field of fisheries science is that, under the new participation proposals, the actual fishing stakeholder finds a way to collaborate in scientific production, while also sharing in making decisions, which replace those imposed by the centralised mechanisms of the past. This enables us to recapture, from a different angle, the link between the debates over collaborative rule-making on the one hand and co-management on the other, both of which, in theory as well as in practice, share the common component of participation in decision-making.

9.3.3.3 Examples of competing construction

There are several cases of competing constructions related to Icelandic Individual Transferable Quota (ITQ) management during the last decade (see, for example, [18]); and there is also a recent case of competing construction derived from a collapse in the anchovy fishery in the Biscay Gulf [19]. An interesting case of competing construction taking account of particular discourses has arrived with the adoption of a moratorium on the Atlantic tuna fishery by the European purse seine fleet. The moratorium has been adopted specifically in the area of the Gulf of Guinea. The generalised use of Fish Aggregating Device (FAD) by the purse seine fleet, along with the intensification of captures with longlines, had signalled probable over-harvesting in some species. This has been of particular importance in the youngest cohorts. The portable FAD is a mechanism that works like an attractor for tuna in the ocean and it constitutes a very effective fishery technique, the use of which was not restricted either inside or outside the 200 miles EEZ. During its intensive utilisation by the European industrial purse seine fleet since beginning of the 1990s, it demonstrated a particular capacity for the capture of juveniles.

However, in 1997, the European purse-seine fleet, through its producers’ organisations (POs), decided to establish a voluntary moratorium, which was implemented and enforced by themselves. This moratorium was in force for three months, between November of 1997 and February of 1998. The other fleets fishing in the same area did not agree to this decision. The following year, the PO proposed to repeat the restriction on the use of FAD and they invited the other fleets to join in with the moratorium, but did not have any success.
Thus, a typical competing-construction debate began. However, there was not an adequate forum of discussion within which to develop the debate and the International Convention for the Conservation of Atlantic Tunas (ICCAT) do not have the authority to compel all fishers to comply with the moratorium. This was despite the fact that, by the following fishing year—when the same fleet proposed to maintain the existing moratorium in the Atlantic in the same terms as the two preceding years and to take a new decision to extend it to the Indian Ocean for a duration of two months—ICCAT produced positive results of the impact of the moratorium, particularly with respect to juvenile cohorts and decided to support seriously the measure.

Subsequently, the EU adopted specific legislation to support the moratorium: EC regulation 973/2001 compels the European purse-seine fleet to comply with the moratorium. However, this regulation does not have sovereignty over non-European fleets, and, thus, this international organisation cannot force affiliated countries to fulfil the terms of the moratorium.

In this case, analysis made by one sector of the harvesting industry has led to a voluntary reduction in captures of juvenile tuna. Other interests in the same fisheries have not agreed with that analysis and have continued fishing. ICCAT have acknowledged the measure, but they have not convinced other fishing stakeholders join their organisation or to comply with this particular regulation. Thus, discourse on this issue remains twofold.

ICCAT recommends the moratorium to its partners. The discourse of the European tuna fleet was constructed, apparently, on a resource preservation basis, and it was a decision taken by a significant element of the private sector on a voluntary basis. This discourse had connections with the philosophical line of ICCAT and other international organisations and councils. Nevertheless, the longline and native coastal fleets did not adopt the moratorium plan. Native coastal fishers argued that European purse seine initiatives have a primarily commercial intent and that their fleet does not hold becoming involved in the sound management of resources as its principal objective. Native fishers remark that, due to an excess in the supply of tuna (particularly yellowfin, skipjacks and bigeye), international prices of these species have been declining since 1999. To avoid losing money and to reach a convenient market adjustment, the challenge for European fleets has been to stop the race fish by, at times, stopping fishing. Coastal fishers argue that the moratorium, on the European fleet, was necessitated by the high level of captures made in relative and absolute terms, over a period of many years. They attribute the saturation of supply in tuna markets to the European fleet and, in the same way, they were also in agreement with the introduction of a long-term severe regulation for FAD. Thus, during the moratorium period, coastal purse seine and bait-boats have increased their captures, arguing that there is a need to develop the domestic tuna fishery in under-developed countries. They support the moratorium for Europeans and suggest the period of moratorium should transferred to increase its effective and restrict the captures of the European fleet even further.

Longliners target older cohorts of tuna species. This kind of tuna have a special market position with sushi-like products, and they have been defending their capture-selectivity discourse. Basically, according to ICCAT data on captures, they have been very active in catches, but less sharply exposed to competing discourse in this debate.
9.3.4 Science as community

Behind the term “competing construction” lies a twofold discourse. Stakeholders are thought, on the one hand, to have interests to defend, and, on the other, to have specific expertise to offer; expertise that is very difficult to obtain by other means. From the point of view of contrasting interests, each has his own perspectives and some of his constructs will compete with those of the rest. At the same time, however, their knowledge of reality enables them to make a potentially very valuable contribution to the knowledge pool. In this context, therefore, it is not hard to appreciate that there are both management-related issues and expertise-related issues at stake [20].

The competing constructs created by the different stakeholders may follow increasingly separate trajectories, leading them into divergent dynamics and eventually to a scenario of conflicting constructions. These dynamics find the scene ready-prepared, with stakeholders from strongly contrasting cultural backgrounds and economic circumstances, non-convergent expectations and an asymmetrical power structure [21]. In conditions such as these, collaboration between scientists and stakeholders will, of course, be of a very elementary nature and the result of the divergence will be seen in poor stock condition.

The whole group of stakeholders may, on the other hand, eventually become aware of the common objectives they share. If so, even while each stakeholder maintains his own position and interests, the community may collectively perceive that areas of agreement do exist. In so far as this is possible, they will see co-operation as a solution that benefits the whole community. It is understood, therefore, that the community as a whole has reached a level of understanding with respect to the need to collaborate to achieve their shared objectives. This situation can accommodate a convergent dynamic between management needs and stakeholder demands, through the pursuit of common goals.

In this scenario, in addition to being affected by regulations, stakeholders also acquire an interest in joining in cooperative tasks. Thus, the stage has been reached in which the whole set of actors has understood that healthy fisheries need to be properly regulated. Stakeholders and scientists alike find the right conditions in which to develop a favourable co-operative dynamic, both in the management arena and that of scientific and technical production. This effectively implies a high level of stakeholder participation in both of these areas. Starting with the competing constructs of the stakeholders, it is worth stressing the collaborative aspect of scientific production, because it is a subject that has not received the same attention in the academic literature as in the research on co-management in business and decision-making.

In this respect, participation is acquiring increasing relevance, because its influence is being transferred both to the decision-making process and to collaboration of stakeholders with scientists and the joint preparation of the scientific research agenda. At the same time, it is becoming easier to establish a framework in which different scientific disciplines can collaborate in order to pool the expertise they have to offer and incorporate it into the process of solving management problems, and, possibly, into the general flow of scientific knowledge [22].

From the point of view of the evolution of the collaboration between the two worlds, such stakeholder-science collaboration is the most worthwhile and most ambitious
scenario for co-operative understanding between the two. It draws from the deference model by accepting that scientists are a necessary tool for a proper analysis of reality, and it tries to integrate the knowledge of the agents in the solution of the scientific and management problems that arise. It also proposes a departure from past histories and competing interests and a switch towards convergent collaboration and the construction of a solid framework for proper resource management.

The fishing community is, without doubt, the fundamental link in the chain. It has the basic characteristics needed to drive this collaboration, because it is a social setting with a shared culture, a close understanding of the goals that can be achieved and a suitable framework to put into practice the agreements that are reached.

9.3.4.1 Examples of community science

In a report developed in the context of collaborative research in fishing industry in the USA, Canada and Europe there is a discussion concerning the intervention of stakeholders in collaborative research in the North Sea [23] The report offers an overview of ongoing collaboration between scientists and stakeholders in several countries of Northern Europe and, subsequently, analyses two Danish experiences. These experiences can be characterised as being close to the community science that we have analysed in this section. The author proposes that these two experiences are like experiments that could be fitted within the community science discourse, and that could be disseminated to advantage. The actual situation is the result of a collaborative dynamic way of developing participation between the stakeholders in the industry. The first project was the ‘Kattegat Sole Project 2004’; and, the second, the ‘North Sea Sandeel Project 1995–2005’. These projects feature a close relationship between the fishermen, scientists and regulators. The projects are, in fact, integrated plans, where collaborations are taking place at different levels and regarding different aspects, beginning with knowledge production and continuing with management and regulative aspects. Fishers take part in a collaborative way in the TAC assessment, they are involved in by-catch evaluation, and they collaborate by completing electronic log-books As Pauksztat [23] argues in relation these projects:

Collaborative research in Denmark seems to be part of a new alignment of the actors involved in fisheries management. Instead of facing each other as potential opponents, fishermen and biologists are increasingly perceived as working together as parties towards the common goal of providing accurate stock assessments and realistic biological advice.

9.4 REGIONAL ADVISORY COUNCILS

9.4.1 Regional Advisory Councils and Participation

Earlier in this chapter we described how participation became incorporated into the models of decision-making. However, the practical results of the theoretical recognition of the importance of participation were, in the area of EU fisheries policy-making relatively limited—consisting of the Advisory Committee on Fisheries and Aquaculture (ACFA)—until Regional Advisory Councils (RACs) recently claimed their place in the system.
The usefulness of RACs is not primarily related to providing the Commission with information on different stakeholders’ positions, although this is also an objective [24], but rather to providing the Commission with advice/opinions, which have already been negotiated from the different positions of stakeholder groups. Even though it is possible to submit majority and minority reports as advice, there is little doubt that consensus-advice will be most influential within the CFP decision-making process. The Commission (and other recipients) will likely be receptive to any proposal coming out of the RACs; but definitely most receptive to advice with broad backing from within the RACs. Consensus-advice has the backing of all the involved stakeholder groups and all groups should ideally feel ownership over measures resulting from this advice. In this sense, the RACs need to arrive at a compromise, like any other decision-making body within Europe.

It should be emphasised that RACs have not been given any decision-making powers. The Commission argued that such a move would not be compatible with the primary legal foundations of the CFP, under the terms of the Treaties of the EU. However, as the following quotation from a high-ranking official in DG Fisheries indicates, there has also been some concern expressed at the European level regarding how RACs would ‘behave’, which might have led the Commission to take the cautious road:

> We certainly have no plan, and I do not legally speaking think we can have a plan, to make these bodies decide anything. I think that their influence could be very great if there is a real discussion in these organisations and a real negotiation and moving away from the sort of primitive defence of short-term interest by everybody. Then I think that the recommendations made from the advisory councils... It would be difficult for the Commission to move significantly away from those. So there may be some quasi-automatic process whereby, even though a proposal has to come through the Commission and go through the Council and the Parliament, we would not normally change what is coming from these advisory councils. If on the other hand they are dominated by fishing interests and purely defend minimum change from the status quo then obviously there will be a problem.

(Interview with high-ranking DG Fisheries official, November 2003, Brussels)

It would clearly be worthwhile to look more closely into why and how the RACs obtained the powers that they did. Equally, it would be interesting to look at how the RACs, as institutions, fit into broader EU developments regarding governance and decision-making procedures. This will, however, not be the focus here.

Alongside the issue of their potential powers, another key problem to be addressed in the preparatory phase leading to the establishment of RACs was to determine what scale the term *regional* should refer to. It is important to note that fishermen deal with the resources on a smaller scale than the regulators, and also that centralised management, taking the whole of the Union as one macro-unit, has not proven effective and is unlikely to do so in the future. The outcome of this discussion was five geographically defined RACs, together covering all EU waters (the Baltic Sea, the Mediterranean Sea, the North Sea, north-western waters, and south-western waters). However, it was also recognised that some fisheries did not fit in to this geographically based scheme and so two regional councils, defined by their type of fishery or species, were also set up: these being for pelagic stocks and for the high seas/long distance fleet [25]. The RACs add complexity to the polycentric, multi-tiered decision-making system, which has developed within the
Union, by introducing a new layer to the system: a level at a larger scale than the Member State, which is recognised as being too low a scale for management of shared stocks; and lower than the EU level, which is at too great a scale to efficiently take into consideration all the different aspects of fisheries in different areas.

In order to obtain legal status, full recognition, and the right to financial and technical support, RACs must adhere to the rules laid down by the Union. RACs are, for practical reasons, limited in their membership, albeit still attempting to guarantee the inclusion of representatives of all the interest groups affected by the CFP. These representatives include those from groups that are directly affected by fisheries decisions—the fisheries sector, which includes the catching sub-sector, shipowners, wholesalers, processors, and women’s groups. These fisheries interests were allotted two thirds of the seats in each RAC. Second, RACs also include representatives of other interest groups affected by the CFP. These are members of society who have an interest in fisheries management but who are outside fisheries interests. These include environmental protection groups and organisations, consumers, and representatives of aquacultural interests and recreational fishermen. Such groups were allotted one third of the seats on the RACs [25].

The EU’s requirement of not discriminating between Member States or stakeholders has meant that inclusiveness has had to be taken very literally: in principle the Greek industry can participate in the Baltic RAC despite no de facto interest in the area. The broad range of stakeholder groups mentioned in the regulation means, also, that there are seats reserved for interest groups who have found it more difficult to send representatives. This has been the case for aquaculture representatives and consumer organisations during the first operational year of the North Sea RAC [26]. Nor are the lines between the two overall stakeholder segments intuitively logical. Anglers and the angling industry could just as well have been put in the industry group, since they are extractive users; and the women’s groups might fit better among the other interest groups, which is actually where the North Sea Women’s Network has ended up on their own request. The anglers opposed this move because, in their view, an industry group is now occupying a seat allotted to other interest groups [26]. In general the grouping of stakeholders into two groups seems technocratic and divisive, and the inclusion of a variety of interests in the group of ‘other interests’ may serve to dilute the voice of the environmental NGOs, who are perceived as the main opposition to the industry, particularly with respect to desired fishing mortality rates.

9.4.2 The knowledge base of the Regional Advisory Councils

From a model of ‘competing constructions’, which we described in theoretical terms earlier, the stakeholders in the RACs will have to move towards a community model and find areas of agreement in the pursuit of properly regulated, healthy fisheries—which is their common goal. Formal scientific knowledge, as well as knowledge in the broader sense, will be very significant in this process. The RACs will advise the Commission on a broad and broadening range of issues and, for this purpose, they are in need of knowledge to act as a lubricant for compromise. In the negotiations over proposals for new management strategies, disputes will often arise over factual issues. This can often be mediated by the input of relevant knowledge, which stakeholders can agree on accepting.
9.4.2.1 Funding and the Limited Pool of Expertise

The RACs have not been supplied with a budget to commission or carry out research of their own. If a RAC wishes to have a study carried out it will have to find external funding for it. This can be done by applying for public funding in the concerned Member States or funding from member organisations. Another way to get the project carried out could be by submitting an expression of interest in the context of the EU’s Research Framework Programmes or other more indirect approaches. The Member States involved in each RAC oblige themselves to provide support to the functioning of the RAC [25]. However, there is no explanation as to what “appropriate support”, which is the wording in the Council [25] regulation, means. It is not likely that Member States will be eager to increase the overall amount spent on managing a sector of decreasing importance. An example of how Member State support could work in practice is the Key Fishing Areas Study.

The proposal for the Key Fishing Areas Study was formulated by the North Sea RAC’s Spatial Planning Working Group during 2004 and 2005. The pilot study aims at mapping key fishing areas in relation to both effort and sensitivity (for example, nursery grounds). Although funding for the entire study remains a problem, RAC stakeholders are expected to receive financial support from the British Department for Environment, Food and Rural Affairs (DEFRA) to conduct a scoping study. This is because DEFRA has set aside an allocation within its science budget to assist the work of RACs. DEFRA, which is a government department with a conservation mandate, recognised, through their participation in the working group meeting, that fish and fisheries risked becoming squeezed by better organised interest groups from other sectors: for example, offshore wind farming. DEFRA, therefore, saw an interest of its own in the study. Important aims of the scoping study are going to be to identify and consider potential sources of funding for the full study as well as making a full project application [27].

The RACs will receive transitional financial support from the EU for their first five years. In the first year the maximum support for a RAC will be 200,000 € (a maximum 90% of the overall operating costs). This figure will decrease to maximum 110,000 € in the fifth year (a maximum 50% of the overall operating costs). In addition, the EU will provide up to 50,000 € each year, with no time limits, to support translation and interpretation (Council, 2004). In relation to the knowledge base of the RACs EU funding only covers “travel and accommodation expenses of experts attending RAC meetings” ([25] Annex II, Part 1). The financial statement for the first half of 2005 for the North Sea RAC shows that, of the total funding of 141,300 € only 16,300 € came from membership fees; and a mere 6000 € was spent on scientific consultants in this period [26, 28]. This indicates that the operating budget will only contribute to a very limited degree to the knowledge base.

The limited funding available for developing their knowledge base puts the RACs in an analogous position with that of DG Fisheries/the Commission itself, which also has a very limited scientific capacity of its own, and largely has to rely on goodwill from the Member States’ national fisheries institutes to provide manpower and support for its Scientific, Technical and Economic Committee for Fisheries (STECF) [24], as well as indirectly with respect to the International Council for the Exploration of the Sea (ICES).
The RACs are, in this sense, going to compete with the Commission (and to some extent also ICES) for the goodwill of Member States and their national fisheries institutes.

Besides the funding problem, there is also the fact that the pool of fisheries expertise is limited. One participant at a meeting of the ICES/NSCFP Study Group on the Incorporation of Additional Information from the Fishing Industry into Fish Stock Assessments (SGFI) in March 2005 reflected over this issue in a discussion:

The RAC must address [...] how scientists can be able to participate in this, there is a great demand on scientific staff, if the RAC wants advice it increases the work load and it will involve three different bodies [STECF, ICES, RACS] giving differing advice on the same issue.

(Observer notes, meeting of SGFI, March 2005, Stavanger, Norway)

In relation to competition for resources, the Commission has already stated its fear that fisheries advisory activities risk being given a lower priority within ICES because the national fisheries laboratories prioritise other work [24]. This effectively means that RACs, if successful in attracting the necessary funding from external sources or members, risk withdrawing scientists from other tasks. However, if the added value of the studies formulated by RACs proves important—more ‘value for money’—then this might not be a problem in the longer term. Nevertheless, it does constitute a potential short-term conflict over limited resources. The conclusion of this discussion are dependent on the RACs obtaining their knowledge base by ‘conventional’ means through the market. An alternative is the suggestion put forward at an SGFI meeting to create a network of “Friends of the RACs”, which would be willing to provide information to the RACs and their working groups in an informal way (Observer notes, meeting of SGFI, March 2005, Stavanger, Norway). This would put less pressure on the RACs and, perhaps, also place them less in direct competition with other actors. It is, however, doubtful that larger scale studies could be carried out in this informal way.

9.4.2.2 Providing Alternative Science and Incorporating Fishers’ Knowledge

If the RACs manage to promote their own research agenda to support their knowledge base, it seems likely that they will promote an agenda where fishermen’s (and other stakeholders’) knowledge has significant weight. However, it is going to be up to the RACs themselves to decide if they should bring forth alternative science, and not merely function as a voice for stakeholder opinions reacting to Commission proposals. The Commission has suggested that the RACs be used for “consultation between scientists and the fishing industry” (section 5.1 [24]), something that seems to be more related to exchanging viewpoints than actually conducting science. Such an approach does not really seem to constitute a move towards ‘community science’ in the sense we have discussed earlier in this chapter. In the same context, the Commission also stated that the RACs will be important sources of information from stakeholders. A participant at the SGFI meeting in March 2005 commented on these ideas from the Commission:

The regulation says very little about the role of scientists, but we have written in that scientists will be invited to participate as experts. They can come from member states, ICES
Participation

or elsewhere, but the RAC will have to pay for its own funds. The Commission seems to think that the RACs are there to give stakeholder views instead of giving alternative ideas, but the RAC wants to give alternative scientific views as well.

(Observer notes, meeting of SGFI, March 2005, Stavanger, Norway, underlined)

This indicates that the RACs will have to do some work to establish a position where it is natural that they advance their own scientific agenda. However, based on the (limited) experience so far, it seems likely that the RACs are determined to do just that. At the moment, at least three proposals for scientific studies, including the Key Fishing Areas Study, have been or are in the process of being developed within the remits of sub-groups under the North Sea RAC [26, 28].

It is explicitly stated in the proposal for the Key Fishing Areas Study that fishers’ knowledge should be taken on board as a potentially very valuable source of information on issues such as “the distribution of fish, their movements in space and time, spawning sites, nursery areas as well as information on fishing activities”. Moreover it is stated that the collection of “fishers’ knowledge is a specialised task, and is not simply a matter of asking questions from a scientific perspective. Much of the information may not fit scientific paradigms” [27]. Much emphasis is, consequently, placed on incorporating fishers’ knowledge.

9.4.2.3 Areas of Interest and Conflicts between Stakeholders in the RACs

In this section we will take a brief look at a few of the issues that can be expected to be of particular interest to stakeholders participating in the RACs. The required knowledge base of the RACs will clearly depend on the priorities of the involved interests, as RACs are mandated to be pro-active and bring issues up of their own accord. However, much of the essential knowledge base of the RACs will of course depend on the issues presented to them. Nevertheless, RACs are likely to have particular hobbyhorses and angles on the different issues.

Furthermore, RACs will, as mentioned above, be most influential on issues where they can come to consensus-decisions. This will clearly not always be easy: lines of conflict will often be between the ‘industry-group’ and the ‘others-group’; in other cases lines of division will also be between industry representatives from different segments or countries. These conflicts are also bound to influence the nature of the knowledge base and how it will be utilised. The following sections draw heavily on Wilson [29].

One of the issues attracting the interest of stakeholders in the RACs is the various kinds of MPAs. The North Sea RAC has already set up a working group to deal with spatial planning, under whose remit MPAs fall. MPAs will be one of the main areas for dispute between stakeholders in the coming years, as the perspectives of the industry and the environmental groups differ. Whereas more or less permanently closed areas are popular among environmental NGOs, these find little sympathy among industry-interests, which instead prefer seasonal closures—aimed at, for example, the protection of spawning stocks—or real-time management—where areas are closed when there is, for instance, too high a by-catch of juveniles, and then opened again when the situation changes. RACs will consequently need knowledge of the effects of various types of MPAs. An important question is whether permanently closed areas are valuable in conserving fish
stocks, which can later be utilised; or rather have only values from the perspective of conserving biodiversity in the marine environment. Information on factors such as bottom type, currents and migrations may facilitate the difficult compromises on issues relating to MPAs. However, it should also be noted that the field of spatial planning and MPAs is an area where the industry and the environmental groups can join forces against other sectors, which want to use the sea or seabed for purposes other than fishing, with likely negative effects for fish, fisheries and the environment: for example, the oil industry and wind farming. The proposed Key Fishing Areas Study is a good example of this. All the stakeholders in the North Sea RAC, as well as DEFRA, have an interest in protecting fish stocks and breeding grounds from the damaging activities of other sectors. Although there are conflicting perspectives among stakeholders on spatial management measures, in practice it has been possible to arrive at the consensus that the knowledge base is incomplete and that this can have negative effects on the position of fish and fisheries activities vis-à-vis the claims of other sectors [27]. It is, thus, both an area of alignment and tension between the different stakeholders.

A cross-cutting issue, which attracts the interests of RACs, is the socio-economic aspects of fisheries management, which is particularly interesting for the industry. This is also going to be an issue in relation to the possible development of MPAs. The North Sea RAC has, as in the case of spatial planning, set up a working group to deal with socio-economic issues. Knowledge about socio-economic considerations is repeatedly mentioned as under-prioritised in fisheries management—ICES deals only with biological and technical aspects of fisheries management—and it is, therefore, highly likely that the RACs will try to balance this by focussing more on socio-economics. Socio-economics is also of particular interest to angling-interests, who believe that the socio-economic value of angling in Europe is underestimated. Angling and angling-interests are going to become increasingly important in the coming years and their active role in the RACs constitutes the first time they have been formally recognised in relation to EU fisheries management.

Finally, technical measures, relating to fishing gear and techniques, constitute an area where the RACs will increasingly contribute and be in need of knowledge. This is an area where scientists conducting a closer relationship with those actually carrying out fishing will be very useful—as has been the case in many cooperative research programmes. The large variety of gears and possible modifications also makes this area particularly appropriate in negotiations where one must arrive at a compromise to become influential. Wilson [29] presents an example of conflicts and conflict-resolution with the intervention of knowledge in the North Sea RAC, which has recently been drafting a multi-annual plan for North Sea plaice. The main part of the plan involved effort reductions through decommissioning and days-at-sea. The impact of 80 mm mesh sizes on certain sub-stocks in certain areas, however, was a sticking point with the industry from just one country, and with the conservationists because of discards. This point was critical and was resolved only when promises were made of further research with respect to mesh sizes and discards in these areas.

The areas discussed above do not, of course, constitute an exhaustive list those issues of interest and relevance to stakeholders in the RACs. Inevitably, the agenda of these
new regional organisations will be driven not only by the stakeholders themselves but, to a large extent, by the requests of the Commission or concerned Member States.

9.5 CONCLUDING REMARKS

The traditional view of public intervention theory placed the regulator in the role of a mere specialist intermediary between collective wills and their technical and scientific transformation into practical public policies. Public action was considered absolutely neutral and technocratic. Thus, social capital and legitimacy was always based on the fact that optimal policies could be developed, using totally objective criteria, in which neutrality had to be guaranteed by the impartiality of the procedures used in the process.

Critics, and their evidence, opposed to the traditional view have generated new proposals to incorporate participation into models in order to explain the public policy-making process. From that time, an inclusive approach has prevailed among the social agents and stakeholders participating in the different levels of decision-making.

The dynamics of the EU’s development have resulted in a multi-tiered institutional model in which the historical institutional structures of the Member States co-exist and interact with those that are being created and developed within the EU. Furthermore, during recent years, the complexity of this structure has reinforced traditional public policy path-dependences. Fishing and agriculture were the first sectors in which the responsibility for public policies was transferred from the Member States to the Union, and they have been regulated at Union level from its earliest stages, essentially in a top-down style.

The knowledge and the quality of the inputs to develop scientific models and applied results, are very important issues when it comes to managing resources in a reasonable way, and this is related to the participation and expertise of scientists and stakeholders. When the views of stakeholders are introduced into the problem-solving process, the scientific formulation of the issues can be elaborated with richer inputs but with new complexities.

In this scenario, in addition to being affected by regulations, stakeholders can also learn to acquire an interest in joining a cooperative task. Thus, to arrive at healthy fisheries they need to be properly regulated. Stakeholders and scientists alike must find the right conditions in which to develop a co-operative dynamic, both in the management area and that of scientific and technical production. This places attention on both sides simultaneously, and it produces a more complex management-science framework.

To put this participation into practice a system of regional-scale, decentralised management units is operational within the EU. They are known as Regional Advisory Councils (RACs). The purpose of the RACs is precisely to bring together the separate elements of management, policy-making, and collaboration in knowledge production into a single forum, in order to develop a co-management model, at a larger scale than the local level and closer to the resource than the unique, but distant, European authority. Knowledge will be a crucial instrument to reaching this destination, and knowledge-based management probably will be a key source of rational resource exploitation.
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Chapter 10

Ecological Side-Effects of Fishing from the Fisheries Management Perspective

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10.1 INTRODUCTION

The effects of fishing on the marine environment can be broadly categorised into direct and indirect ones. Direct effects include: (i) mortality of target and by-catch species; (ii) changes in food availability and competition for resources among certain predators, such as marine mammals, large fish and sea birds; (iii) discards of unwanted catch and fish offal and (iv) physical and biological disturbance of the seabed (benthic habitats and benthic species). Indirect effects include: (i) changes in the genetic composition of fished populations and communities; (ii) changes in species and size composition of the marine communities; and (iii) changes in energy flow in the ecosystem. Environmental impacts of fishing can be either short-term, or long-term with more irreversible effects, the first ones of which being more easily evaluated.

Evaluation and management of the side-effects of fishing demand:

• Definition of what is an operational ecosystem and what is an operational ecosystem-based side-effect of fishing. Spatio-temporal and trophic-level scaling, of all system components (including habitats, stocks, communities, bio-diversity, fisheries, etc.), is essential here.
• Formulation of specific and operational management objectives for side-effects of fishing which are generally agreed upon and have a common understanding among stakeholders. This should be done in agreement with the existing and prevailing general (overall) management objectives of sustainability and the precautionary approach and its principles.
• Definition of the exact management action needed to be implemented, in order to reach the specific management objectives.
• Definition of parameters (metrics) or key indicators that can and should be measured to evaluate the ability of the adopted management measures to achieve the pre-defined objectives. Target or limit reference points for those parameters and key indicators should be established and risk analysis methods and uncertainty evaluations should be developed to evaluate the consequences of potential failures to meet these points.
• Definition of adequate data levels for the parameters and indicators as well as uncertainty levels for these, and development and maintenance of systems to monitor, evaluate and develop advice on the ecosystem components and their functioning, as well as the fishery impacts. It is important to be able to distinguish between anthropogenic effects and those due to natural and environmental factors.

Although, our ability to fully manage marine ecosystems in a sustainable manner is rather limited due to their high degree of complexity and, at best, we are only capable of managing first and maybe second-order impacts of our own activities, adoption of much more holistic (ecosystem) management approaches would be beneficial in addressing the ecological side-effects of fishing. Such approaches, however, need to have specific, clear and widely agreed management objectives in order to be effective [1]. Because of this complexity, less holistic approaches to formulating specific management elements and objectives, in relation to small-scale and local ecosystem side-effects of fisheries, are ongoing in relation to OSPAR and the Bergen Declaration [2, 3].

In the subsequent sections we summarise the main ecological side-effects of fishing and discuss various management approaches and actions for facing these effects.

10.2 MAIN ECOLOGICAL SIDE-EFFECTS OF FISHING

10.2.1 By-catches and discards

On average, a total of 110 million tonnes of marine animas are removed annually from the world’s seas through the various fishing operations. These include both commonly targeted commercial species and those other species that are caught, although they are not targeted (by-catches). Certain by-catch individuals are retained for sale, but, most often, they are thrown back dead (discards), as they include species of a low economic value, endangered and/or protected species, undersized individuals of valuable commercial species, or they are simply surplus to the fishing operation’s quotas. The United Nations Food and Agriculture Organisation (FAO) reports that discards represent, in terms of biomass, as much as 25% of the total world catches [4]. In several cases, discards influence the assessment of stock status and dynamics, affecting the fisheries management of both target and by-catch species, e.g. North Sea plaice and haddock fisheries [5–7].

A great number of species are included in the discards, many of which are bony-fish and sharks. Fisheries have a greater impact on discarded species that cannot survive (for example, species with swim-bladders). However, the redirection of large amounts of energy into marine ecosystems through discards can also lead to community structure changes in scavenger species [8].

By-catches of protected species, such as marine mammals and sea-turtles, occur in several fisheries. Turtles are particularly vulnerable to entanglement and drowning in gill nets and associated ghost-fishing gears, as the rough skin on their head and flippers catches easily on the meshes of these nets [9]. Depending on the area, shrimp trawl fisheries are a significant source of mortality for turtles [10].
The eastern Pacific tuna purse seine fishery, which began in the late of 1950s and replaced the highly selective pole-and-line fishery that had operated previously, had large by-catches of dolphins. Crude estimates of the number of dolphins killed during the 1960s indicated a mortality rate of hundreds of thousands per year, and this caused a population decline until the late 1970s. This sparked the “tuna-dolphin debate” and resulted in the “back-down procedure”, which is a process to release dolphins from the net, and the provision of independent observers aboard tuna vessels to record dolphin mortality [11]. Gill nets used to catch swordfish and tuna, although highly selective with respect to the size class of animals captured, are associated with high numbers of incidental captures of cetaceans. Also seals are entangled in nets or buoy ropes [12].

Seabirds are also caught incidentally in some fisheries. In general, set net, driftnet, and long-line fisheries take the most incidental catches of birds. Seabirds are caught by invisible underwater nylon nets while diving to catch small fish and their mortality can be particularly high if nets are deployed in close proximity to breeding colonies. In some circumstances, bird deaths associated with nets can constitute the greatest source of mortality for some local populations. Birds are also attracted by baited hooks, as they are paid out from long-line vessels, or if they can see bait at shallow water depths. There are both conservation and financial incentives to reduce bird by-catches in long-line fisheries. For example, seabirds in the NE Atlantic were reported to remove 70% of the bait from long-lines [8].

### 10.2.2 Effects on the population level (changes in size structure and life history parameters)

Variations in vulnerability to fishing and the selectivity of fishing gears affect the size and age structure of fished populations. In many ecosystems worldwide, the composition of the fish fauna has changed from large fish-eating fish species towards smaller plankton-eating fish species [13, 14]. For instance, the ability of fished species to avoid trawl-nets depends on swimming speed and escape responses. Small fish are more vulnerable to trawl fisheries as swimming ability is related to body length. Fishers, however, target mainly the larger fish using size selective gears (mesh size of nets, hook size, etc.). In general, intense fishing has resulted in most, if not all, cases in significant abundance reductions of larger individuals, gradually leading to “growth overfishing” conditions. These changes in the size and age structure of a population can influence various aspects of the life history (for example, growth rate and fecundity).

Size selective fisheries can change the sex ratios of fished populations and curtail reproductive life span. In hermaphroditic species, sex change occurs at a critical size, so fishing can bias their sex ratios [15–17]. In protogynous species, such as porgies and groupers, changes in the sex-ratio may be precipitated by fisheries removing the larger males. For North Sea plaice, sexually dimorphic growth, gear selectivity and discarding cause changes in stock sex ratio. As a consequence of this, assessment of the stock and stock perception becomes more biased towards the female part of the population with increasing fishing mortality resulting, if discards are not taken into account in the assessment [18]. Changes in sex ratios have been also observed in exploited invertebrates [19]; the effects of these changes, however, have scarcely been examined.
Reproduction may also be affected by shifts in the size and age distributions of fish populations. The relative fecundity (number of eggs per unit of body mass) of fish increases with body size, and, thus, the population of a given biomass has greater potential fecundity when composed of larger, rather than smaller, individuals. There is evidence that when reproductive life span is artificially curtailed by fishing, fish may compensate through changes in life history parameters. This has been observed in the case of the heavily exploited North Sea stocks of plaice, cod and sole, where changes in growth, maturation and fecundity have compensated for about 25% of the losses in total egg production [20]. Likewise, for the Mediterranean wrasse *Coris julis* (Labridae) population, Harmelin *et al.* [21] reported an earlier sex change resulting from modifications in population size structure induced by heavy fishing.

Decrease in population density following fishing may lead to increased growth rates, because lower population abundance should lead to increased food availability for the remaining individuals. Growth rate changes may be accompanied by other changes in life histories to compensate a curtailed life span due to fishing. Such a change has been observed in the heavily exploited Norwegian spring spawning herring which, in the 1970s, matured at 2–3 years earlier than it had in the 1950s and 1960s, when the stock abundance was higher [22, 23].

**10.2.3 Effects on species diversity and genetic structure of communities**

Fishing is, for the most part, selective, and has, therefore, the potential to introduce changes in the genetic composition of exploited stocks. The selection differential is the difference in the value of a particular trait with and without fishing. So far, we know too little about the relative contribution of phenotypic plasticity and genetic change in the response of life history characteristics to fishing. [24].

Large-scale patterns in fish diversity are primarily governed by bio-geographic factors [25, 26]. These patterns are modified by the direct and indirect effects of fishing. A community of high evenness and low dominance is considered more diverse than one with the same number of species but low evenness and high dominance (where few species account for most of the total abundance). Changes in the relationship between species number and abundance over time, or comparisons among fished and un-fished areas, may indicate the effects of fishing on community structure.

Trawl surveys of the North Sea demersal fish community provided evidence for changes in diversity due to fishing, showing that the community has been dominated by fewer species in recent years [27]. Species diversity declined in the areas where fishing intensity was highest, but fishing effects were largely confined to decreases in abundance of species with slow life histories, which are particularly vulnerable to exploitation [28]. This suggests that changes in the abundance of vulnerable indicator species, such as sharks and rays, rather than broad measures of community diversity, provide a better measure of fishing effects in this case. Fishing effects have been more dramatic in ecosystems where few species fulfil key functional roles.

Species extinctions and extirpations due to fishing also influence ecosystem diversity. Extinction is the permanent loss of a species, while extirpation refers to local losses of stocks or subpopulations. Fishing has extirpated many fished species on a local scale
and extirpation is likely to reduce genetic diversity. Factors making species vulnerable to extirpation by fishing are: limited geographical distributions; dependence on specific habitats; slow-life histories; and accessibility to fishers [8]. Although extirpations may lead to extinction, the probability of biological extinction of a major target species is rather low as it is likely that its fishery will pause early enough due to a lack of profit. However, this is not valid for certain by-catch species such as skates, which can only tolerate low levels of fishing mortality due to their low fecundity, large body size and slow life history. In the Atlantic, for example, they suffer high fishing mortality because they are caught by trawlers chasing primary target species such as cod and haddock [29–32]. As a result, four of the larger species of the North Sea skates and rays are currently fished at levels that may extirpate them, as they are all vulnerable to groundfish trawlers well before they attain maturity [8].

Life history traits such as age and size at maturity, growth rate and reproductive output have a genetic basis, and selective predation on fishes by other fishes is seen as a major cause of evolutionary change [33, 34]. In experimental systems, selective harvesting has led to genetic changes. Exploited populations will evolve in response to harvesting; although it has often been difficult to detect heritable responses to exploitation because they are masked by phenotypic effects [35]. In the case of the North Sea plaice, Pleuronectes platessa, however, where age and size at maturity have fallen during intensive exploitation, at least part of this fall has been attributed to genetic effects [36]. Similar effects may have also contributed to size at first maturity decreases of North Sea cod during the last 100 years.

10.2.4 Other effects on community structure

On the global scale, fishing has affected the structure of marine ecosystems. Fish communities pass through a series of structural changes as they are increasingly heavily fished. At first, larger individuals of all target species decrease in abundance, and larger species form a smaller proportion of the total abundance [37]. Ultimately, the whole community becomes dominated by smaller individuals and smaller species [13, 14].

There are many examples of changes in community structure, particularly on tropical reefs [38–41]. In the North Sea, there have been marked changes in the relative abundance of species since trawl surveys began in 1925 [27, 42]. A phylogenetic analysis showed that species suffering from higher abundance decreases than their nearest evolutionary relatives were characterised by greater size at first maturity and slower growth rates towards a greater maximum size. They also had lower rates of potential population increase. The susceptibility of late maturing and larger fishes to fishing suggests that small and early maturing species would increase in relative abundance in an intensively exploited multispecies community. While the life histories of smaller species may enable them to sustain higher instantaneous mortality rates than larger species, they may also suffer lower fishing mortality simply because they are less desirable and escape through meshes in nets and traps.

Changes in species’ relative abundance are also reflected in the trophic structure of communities. In general, species with faster life histories feed at lower trophic levels and have higher production biomass ratios. On intensively fished reefs, fish biomass is
dominated by herbivores, while invertebrate feeders and piscivores dominate on lightly fished reefs [40]. The composition of target fish communities can change rapidly in response to fishing. Changes in trophic structure observed in individual fisheries are also reflected in global landings. Thus, the mean trophic level of marine landings fell between 1950 and 1993 as fishers “fished down the food chain” [13]. In heavily fished areas, such as the north-east Atlantic and Mediterranean, there was a consistent downward trend [8].

Another change in global landings that may be a response to the abundance reduction of large predators is the increased landings of predatory cephalopods. It has been suggested that these species are more abundant, and, therefore, more likely to support fisheries in ecosystems where fish predators are overfished [43]. Cephalopods are likely to sustain relatively high fishing mortality due to their fast life histories.

10.2.5 Impacts on benthos and benthic habitats

Fishers, and/or their gears, interact with the habitat to some degree. The interaction of fishing gears with the environment either disturbs the habitat directly (physical disturbance), or indirectly, by removing competitors and predators from the system (biological disturbance).

The severity of disturbance can range from damaging only the most sensitive organisms to destruction of all multicellular life [44]. There are a number of interacting factors that determine the impact of a disturbance in a particular environment. These factors include habitat stability (for example rocky vs. loose and sandy substrata), frequency of natural disturbance (related to depth, exposure and current regimes), the type of fishing gear used, and the scale, intensity and frequency of fishing activities [8].

As fishing activities are not homogeneously distributed in space, but rather concentrated in the most productive areas (for example, continental shelves), knowledge of the distribution of fishing effort is essential to predict the likely consequences of fishing on benthic habitats and communities.

10.2.5.1 Physical disturbance

Active fishing techniques, such as towing trawls, dredges, chemicals or explosives, cause serious physical disturbance by the direct contact of the gear with the seabed. In soft sediments, this will lead to the turbulent re-suspension of the surface sediments which may remobilise contaminants and expose the anoxic lower sediment layers. On hard substrata, boulders may be physically moved or rock reef and biogenic structures may be destroyed [45].

The magnitude of impacts on the seabed is determined by the towing speed, the physical dimensions and weight of the gear, and the type of substratum. The resultant changes to the habitat may persist for only a few hours in shallow waters with strong tides and continuous wave action, for several years on muddy sediments found in sheltered areas such as fiords, or for decades in the relatively undisturbed deep sea [8].
10.2.5.2 Biological disturbance

Bottom trawling and dredging affects infauna (animals living entirely within the sediment) and cause less common species to be more severely depleted [46]. Such fishing methods can also reduce the patchiness of benthic communities. However, there are no easily detectable effects in habitats exposed to high level of natural disturbance (mobile sand habitats). Such habitats tend to be dominated by small-bodied opportunistic fauna (for example, spionid and capitellid polychaetes and amphipods), which rapidly recolonise areas that have been physically disturbed [47, 48]. At larger scales of disturbance, recolonisation probably takes longer. Generally, in mobile sedimentary environments larger species, especially bivalves and sea urchins, may be the most vulnerable to trawling disturbance [49]. As sedimentary habitats become more stable, the effects of fishing disturbance are more dramatic in the short term and last for longer. This principle applies both to the structure and composition of the benthic assemblage and the topography and physical structure of the sediment.

Changes in epifauna induced by trawling and dredging may have severe effects on benthic communities, as several examples demonstrate:

- Colonies formed by the tube-building polychaete worm *Sabellaria spinulosa* in the European Wadden Sea that eventually become reefs if left undisturbed, have been destroyed by years of dredging and trawling activities, and replaced by small polychaete species (this is a typical response of benthic communities to stress) [50].

- A typical feature of undisturbed benthic communities is the dominance of bryozoans, hydroids and tube worms, which increase the three dimensional complexity of the habitat; thus providing shelter for juvenile fishes, reducing their vulnerability to predation [51]. Areas with intense fishing activity have lower species diversity, reduced habitat complexity, and are dominated by species resistant to fishing activities: hard-shelled bivalves, echinoderms and scavenging crabs.

- Maerl beds are formed from calcareous algae of the genus Lithothamnion, which are amongst the oldest living marine plants in Europe. These beds can take hundreds of years to accumulate, with live material at the surface of the bed overlying dead thalli [52]. The branched structure of the thalli provides unique and complex habitat, which supports a diverse community of animals, some of them commercially important. Dredging severely disturbs this habitat, causing long-term changes to the composition of the associated benthic fauna [53].

- Seagrass meadows are highly productive, support complex trophic food webs, and act as breeding and nursery areas for species of commercial importance. Seagrass meadows are vulnerable to physical disturbance by bottom-fishing gears that tear up individual plants, reduce biomass by shearing off fronds, and expose rhizomes. Destruction of parts of seagrass meadows reduces the stability of this habitat [54–56].

10.2.6 Ghost fishing

Static gears, such as pots and nets, can be lost as a result of trawling or bad weather conditions. These gears, or parts of them, could continue fishing for a long time. This phenomenon is termed "ghost fishing". It is not simple to make general statements about the
longevity of ghost-fishing gears, or their ability to continue fishing, based on experimental studies. For most lost nets, over the first few days catches decline almost exponentially as the increasing weight of the catch causes the net to collapse. Then, for the next few weeks, the cadavers within the net attract large number of scavenging crustaceans, many of which are valuable commercial species and also become entangled in the net. After this initial period, the cycle of capture lasts as long as the net has some entanglement properties [57, 58].

Pots or traps are likely to maintain their shape, and, therefore, capture efficiency, for much longer than ghost nets. Ghost-fishing mortality rates of up to 55% of the mortality rates recorded in attended pots have been reported [59, 60]. Little is known about the frequency of net or pot loss. The few available estimates of gear loss indicate that it can be substantial in some fisheries [61–64].

10.2.7 Interaction between seabirds and fisheries

Interactions between seabirds, mammals and fisheries occur as they may compete for the same prey species in the same areas. In upwelling and productive coastal systems, seabirds can consume 5–30% of fish production. Seabirds prefer small pelagic fishes, which are relatively easy to catch and rich in energy. Various studies show consistent linkages between prey availability and reproductive success of seabirds. For example, puffin (*Fratercula arctica*) and murre (*Uria* spp.) populations declined with the collapse of the Norwegian herring *Clupea harengus* and Barents Sea capelin *Mallotus villosus* stocks [8]. While, industrial fisheries and oceanographic changes in the North Sea seem to have had a combined effect on populations of North Sea black-legged kittiwakes on a local scale [65].

Fishery discards also have an impact on seabirds [66]. Much discarded material floats on the sea surface, or sinks slowly. Seabirds follow fishing boats and consume discarded material. The effects of discarding, particularly on the scavenger seabirds, are reflected in their population trends—increasing for many of these species.

10.3 SELECTED EXAMPLES OF EU FISHERIES HAVING SIGNIFICANT ECOLOGICAL SIDE-EFFECTS AND THE RELATED MANAGEMENT ACTIONS

10.3.1 Discards in EU fisheries—the case of the North Sea Flatfish Fisheries

Discarding practices are common in all important EU human-consumption fisheries. Approximately one quarter of the global catch is discarded annually according to the FAO. The effects of these discards are complex and vary from fishery to fishery and from year to year. Consequently, it is impossible to provide a comprehensive description of all their effects. For most fisheries, the existing level of information on the species and size distribution of discards is very limited due to low sampling intensity and inadequate fleet coverage. In addition, information provided by the commercial fisheries themselves, regarding the total discarded quantities, is rather poor. As a consequence, management
and biological advice can rarely be satisfactorily adjusted to take into account the effects of discards on stock dynamics and on the ecosystem.

If removals of target species in a fishery are larger than can be perceived from landing statistics and discard rates, and especially variations in those, and this is not accounted for in assessments, then stock evaluations, as well as predictions, may be seriously flawed, resulting in uncertainty and possible bias in the scientific advice. Thus, discarding may directly influence the effectiveness of the existing management system. Typically, poor or missing estimates of discards introduce uncertainty and bias in recruitment estimates, thereby affecting scientists’ understanding of stock–recruit relationships and, consequently, the management reference points derived from them. Moreover, discarding has a wider ecosystem impact, as well as causing an unknown level of mortality on a large array of apparently non-targeted and unexploited species, it also affects the available food levels for scavenger species.

Accounting for discards is becoming a common request for fleet-based management advice, and is particularly crucial in assessments and predictions related to mixed fisheries. The current EU Data Collection Programme comprises actions to estimate discards for various stocks and major fisheries. Although until 2007/08 sampling is scheduled on a country and species basis, (i.e., not on a fleet basis), and, therefore, not fully covering all the important fleets and fisheries, it is expected that our knowledge on discard quantity and composition will be improved in the coming years. However, as the collection of data on discards is expensive, due to the typically required on-board sampling, it is necessary to find a compromise between cost, sampling coverage and precision estimates.

Below, we outline the example of the North Sea flatfish fisheries, where records of discarding practices are comprehensive, and where effects on the stocks, as well as on the management of the related fisheries, have been documented and evaluated.

In the North Sea, annual fish captures represent about 30% of the total fish biomass. Discarding has a major impact on the ecosystem and fisheries, as more than 30% of the catch is thrown overboard [5]. Most of the discarded volume consists of small fish and invertebrates.

The beam trawl fishery in the North Sea, targeting mainly sole and plaice, is an example of a mixed fishery with extensive discarding practices. Compared to other fisheries employing towing gears, such as ordinary bottom trawls and seines, beam trawls have much higher discard rates. Apart from valuable by-catches of turbot and brill [6], a range of other species may be landed or discarded (for example, dab, flounder lemon sole, grey gurnard, cod, and whiting). The minimum legal mesh size when fishing for sole was 75 mm until 1987 and 80 mm thereafter. This has been set in accordance with mesh selection characteristics for sole. The beam trawl fishery is, to a high degree, conducted in areas with a high abundance of juvenile plaice. As a consequence, large numbers of young plaice are caught and discarded, resulting in marked impacts on the perceived stock dynamics of plaice, and on stock level forecasts [5–7, 18, 67]. This discarding occurs on the basis of minimum landing size regulations for plaice. Discard rates are affected by variations in growth rate, recruitment, and relative distribution of the fish and the fishery. Apart from plaice, discards include many other fish species, as well as large quantities of benthos. For instance, dab is the most abundant species in the catch, but, due to its
Historically, plaice discard rates have varied in response to market conditions and changes in minimum landing size. Nevertheless, quantitative estimates have been sparse [7]. Clear seasonal and spatial patterns have been identified, with discards being higher during the second and third quarter than the rest of the year, and also higher in coastal areas than offshore [6]. Further, van Keeken et al. [5] observed high variation in plaice discard percentages in different seasons and fishing grounds.

Recently, van Keeken et al. [5] produced a time series of discard estimates based on samples, simulations of the fraction of sizes of each cohort that may escape through the meshes, and the fractions of marketable and unmarketable plaice retained by the gear. They also took into account historic changes in growth rate and changes in availability to the fishery (based on survey data). This was compared to the minimum landing sizes for plaice. The resulting estimated percentages of plaice discarded (size range 17–28 cm) ranged from 30–80% and showed substantial inter-annual and between-cohort variations.

Kell and Bromley [18] reconstructed a discard time series based on cohort growth rates, gear mesh size and sorting behaviour of the fishery. The proportion of each age group in the discards/landings fraction was calculated, and, subsequently, the fishing mortality rate, obtained from VPA based on landings, was corrected to include the discard mortality, using equilibrium age-structured production models that combine SSB-per-recruit, yield-per-recruit and the stock–recruitment relationship. The study evaluated the effect of sexually dimorphic growth, overall recruitment and growth changes, and density-dependent sexual maturation on discard rates, and considered their implications for the current assessment and management practices, as well as for stock productivity. The conclusion was that discard mortality of small plaice has a substantial influence on the estimated productivity, recruitment and stock size of the species, which, in turn, have important management implications. One major effect of not including discards in the assessment was that the potential gains in yield by reducing $F$ below $F_{pa}$ were under-estimated, because the corresponding reduction in discard mortality, enabling more fish to survive and grow to a landable size, was not accounted for. When there are large changes in stock productivity over time scales of 10–15 years, as those observed for North Sea plaice, our perception of stock status, relative to limit and target reference points, will change if discards are included. However, the method applied by Kell and Bromley [18] is problematic when estimating $F$ on age groups from which only a few fish above the minimum landing size are landed. Also, the method does not allow for variations in resource availability caused by changes in the distribution pattern of fish and fisheries [5].

Until 2004, discard information on target species was not included in routine ICES (International Council for Exploration of the Sea) plaice assessments, because reliable annual estimates of discards-at-age were not available [5, 67]. Despite various shortcomings, in 2004 and 2005 ICES accepted a new single species assessment that included discards, which was based on the work of van Keeken et al. [5]. The biological target and limit reference points were accordingly changed.

The ecological impact of the large quantities of discards produced by the beam trawl fishery in the North Sea is not well-known, but discarding provides additional food for
scavengers [6]. Studies by van Beek [6] and Fonds [68] indicate that more than 90% of the individuals of most fish species are dead, or eventually die after having been returned to the sea. A relatively small proportion of all discards may be utilised by seabirds: Camphuysen et al. [66, 69] estimate that about one third of total discards are consumed by seabirds; where 95% of discarded offal, 80% of roundfish, 20% of flatfish, and 6% of benthic invertebrates are consumed by scavenging sea birds (see also [70, 71]). The rest will sink to the sea bottom and serve as food for benthic scavengers, including fish. Although scavengers may be favoured by discarding, this might be at the cost of predators that prefer to take live prey and are faced with less available food. However, this applies to all catch, whether landed or discarded. Studies, on the spatial distribution of Dutch beam trawl effort, indicate that 70% of effort is exerted on 20% of the total area visited by the vessels participating in the fishery [72]. This indicates that the major part of the discards occurs in a relatively small area, with ecological consequences being largely unknown.

General management objectives aiming to reduce discards in mixed fisheries have been set by the EU through, for example, the European Commission Green Paper. Additional objectives aiming for the protection of the environment, including discard aspects, are in the process of being formulated through the identification of Ecological Quality Elements for the North Sea—specifically the ECOQ Element (b): namely, “Presence and extent of threatened and declining species” [2, 3]—and followed up by a process of development and formulation of specific ECOQO’s (Ecological Quality Objectives) through ICES [2, 3, 73, 74]. However, so far no specific and operational ecologically based management objectives have been defined; nor have limit reference points for non-target species in the mixed human-consumption fisheries of EU waters.

The North Sea mixed flatfish fishery is one of the few examples of the existing knowledge basis of ecological side-effects of fishing being used. Here, discards of one of the two target species in a fishery are taken into account in the assessment of the stock and, consequently, in the formulation of biological advice and specific management reference points for sustainability.

For the non-commercial species of the fishery there are no assessments or reference points to take into account the ecological effects of discarding on those species. Only regulations relating to the selectivity of fishing gears (for example, minimum mesh sizes) are implemented, which aim to minimise discards, but do not have any specific management objectives. Consequently, from the overall available knowledge on the potential ecological side-effects of discards, only information on plaice discard rates, as opposed to discards of other species, is considered when formulating specific management objectives and reference points for managing this fishery.

10.3.2 North Sea beam trawl fishery—effects on the benthic community

The North Sea beam trawl fishery has important impacts on benthic habitats and fauna. These, both short and long-term impacts, are, to a large extent, well-known and extensively described in a wide variety of literature and reviews of scientific knowledge. However, in this context, it is also known that the large-scale effects of anthropogenic factors are difficult to distinguish from the many influential natural and environmental factors in
the very complex marine ecosystems. Consequently, the various effects of beam trawl fisheries on the North Sea benthos will only be summarised here, as the main aim of the section is to discuss the utilisation of this knowledge in the management and evaluation of the fishery and ecosystem.

Impacts are the product of physical and biological disturbances by fishing activity, and the sum of effort exerted on a local spatial scale. Towed fishing gears, operating in contact with the seabed, annually sweep an area corresponding to the size of the North Sea. The distribution of effort from fisheries with hauled gears is typically very patchy [75], which is also the case for the North Sea beam trawl fishery described in the previous section [72]. Accordingly, the spatial scale of the impact is central in evaluation.

The biological, physical and geo-chemical implications of the re-suspension of sediment caused by fishing activities are only poorly understood but may be significant [24, 76]. In the case of the beam trawl fishery, the tickler chains cause major physical disturbance and damage to the fauna, despite attempts to improve gear design [24, 77, 78].

Fishing redirects energy within the marine ecosystem in two ways: (i) animals killed or displaced on the seabed become available for consumption by predators and scavengers; (ii) material brought on board fishing vessels and discarded becomes available to avian and mid-water predators and scavengers [24].

The mean initial response and reduction in benthos to physical disturbance by bottom gear is found to be significant by Collie et al. [79], and they conclude that the direct impact of scallop dredging is very obvious. Through the analysis of historical records concerning the impact of fishery and environmental factors on the benthic community of the North Sea, Frid and Clark [80] concluded that the greatest changes in the benthic fauna associated with bottom fishing have probably occurred some years ago with the removal of long-lived and slowly reproducing and reef-forming fauna [24]. Life history characteristics of large size benthic animals, such as slow growth rate, and late age at first maturity makes them more vulnerable to fishing disturbance [81]. Absolute annual fishing mortality rates of benthic megafauna (>1 cm) in sandy sediment has been estimated to range from 5% to 39% (and for some species even more) as a result of beam trawling in the North Sea [82]. Muddy sediments of a stable nature are more sensitive to disturbance, and, in the long term, more susceptible to fisheries than coarser and more dynamic sediments. Visible physical effects and reductions in species diversity and abundance of certain species on muddy sea-beds were in evidence more than eighteen months after the disturbance [83]. The recovery rate appears to be more rapid in less stable habitats, which are generally inhabited by more opportunistic species [79]. Habitats constructed by, or composed of, living organisms are likely to take a long time to recover from disturbance by hauled demersal fishing gears—for example, maerl beds. These, and other studies, suggest that chronic fishing disturbance has altered North Sea benthic communities over time, even though many of these types of studies are hampered by the absence of reference areas and suitable control areas that have not been influenced by fisheries [24].

In the North Sea, discards and damaged biological material from beam trawling for plaice is rapidly consumed by opportunistic scavengers. Trawling results in increased rates of recycling of macro-benthic fauna and fish through the food web, and, to some extent, increases food subsidies and the growth of some fish species. The annual amount of food supplied by beam trawling has been found to be approximately 7% of the maximum
annual food demand of all common benthic predators together [84]. In addition, trawling has changed the abundance and size structure of some populations of predatory fish, their feeding habits [80], and the species diversity in benthic communities, especially on stable substrates [85].

For the North Sea, there are several single study examples of ecosystem effects of fisheries, even though available data and historical time series data for these are scarce, and of limited quality, considering the complexity of the ecosystem that is described. Scientifically rigorous and quantitative evidence on large temporal and spatial scales is difficult to obtain [79, 80]. Thus, there are clear uncertainties surrounding the current knowledge and ability to predict wide-scale effects of fishing and the marine environment [24].

General objectives in, for example, the precautionary approach, call for the need to integrate fisheries and ecosystem management. So far, current management only takes into account a few obvious irreversible impacts of fishing—for example, by protecting some species that are at risk of extinction—and specific management objectives have only been implemented in relation to maximum by-catch levels, and not in relation to discards. Management strategies and methods to provide protection to the wider ecosystem and its functions are not well developed [80]. This would imply that, in addition to numerical management, parametric management, that considers the wider physical and biological parameters of fishery, is also essential [24]. Even though ecological consequences and impacts of fisheries are well recognised and documented, it has proven complicated to identify “global” indicators of these effects and to identify ecologically based specific reference levels and limit reference points to be used as a basis for formulating operational management objectives. Thus, no holistic approaches, in the form of ecosystem-based management, have, so far, been implemented in relation to the inclusion of side-effects of fisheries on the wider ecosystem. Only isolated management measures exist, such as those concerning gear selectivity and by-catch limits, which offer limited protection to the ecosystem against the adverse effects of fishing.

Through the ECOQ Elements identified by OSPAR and ICES [2, 3], attempts to formulate specific management objectives (ECOQOs) and reference points in relation to the side-effects of fishing on the benthic community are ongoing. These cover: (i) threatened and declining species with respect to the presence and extent of threatened and extent of these in these North Sea (with the objective of implementing species recovery plans for all threatened and declining species, although no specific limit reference points are proposed by ICES); (ii) fish communities, with respect to changes in the proportion of large fish and, thus, the average weight and average maximum length of the fish community (although, again, no specific objectives and limit reference points are proposed by ICES); (iii) benthic communities with respect to imposex in dogwhelks (where objective and limit reference point of a reference level for TBT concentration (and imposex) is zero (Vas Deferens Sequence Index <0.3) and a limit point of VDSI >5 are proposed by ICES); (iv) density of sensitive (for example, fragile) species, and density of opportunistic species (ICES recommends that OSPAR consider dropping this ECOQ element as these species are ubiquitous and provide no close link to anthropogenic activities); and (v) restore and/or maintain the extent of threatened habitats (in this respect ICES recommends that features of flat oyster beds, intertidal mudflats, and littoral chalk communities should be further developed as a basis for an ECOQO for this revised ECOQ element,
and ICES recommends that features of two other threatened and declining habitats in the North Sea—seapen and burrowing megafauna communities and seagrass beds—should not, at present, be used as a basis for ECOQOs) [2, 3]. None of these ECOQOs and corresponding reference levels and limits points have been adopted and implemented as yet.

10.3.3 Industrial fisheries for sandeel in the North Sea—effects on food resources for seabirds

Most North Sea seabirds have expanded their breeding range, established new colonies, and have increased in number over the last century [70]. Over the last 20 years or more, the growth of some of these populations has ceased, but seabird numbers in the North Sea are generally at a historically high level [65, 70, 86]. It was previously thought that increases in population size of many avian scavengers were largely related to increases in the amount of fisheries-generated waste [24]. In general, up to 30% of the food of some seabird species is sandeel and up to 30% is discards. However, Camphuysen and Garthe [70] emphasise that seabirds remain reliant upon natural sources of food that are vulnerable to over-exploitation. Fisheries wastes may have aided the expansion of some species by providing a source of food when alternatives were unavailable. Furthermore, reduced harvesting of seabirds as well as complex trophic interactions from reduction in populations of large piscivorous predators (gadoid fishes mainly) in the North Sea may also have positive effects on seabird populations. However, sudden changes in the distribution of fishing effort harvesting bird-prey fish species may have unexpected side-effects for seabirds. The interactions between industrial fisheries and important predators, such as the black-legged kittiwake, with respect to food availability are not well known [2, 70].

Like other nesting seabirds, black-legged kittiwakes largely ignore discards as a food supply over most of the breeding season, preferring to capture sandeels and other small fish. It has been shown that several species experienced poor breeding success when proportionally large amounts of discards featured in chick diets [70]. Long-term changes in populations of animals as a result of fishing are more clearly seen in large-bodied organisms such as birds and certain fish species [24]. Many top predators (birds, cetaceans and seals) rely on sandeels as the main part of their food [2, 87]. Even though seabird populations, such as kittiwakes and arctic terns, have increased during this century in parallel with fishing, and fishing may have benefits for seabird populations, concerns have been expressed that industrial fisheries may have severe local effects on certain seabirds, by depleting feeding grounds in their breeding area.

Catches by industrial fisheries of small fish, including sandeels, sprat, herring and Norway pout, in the North Sea have increased significantly over the last 50 years [67]. This exploitation of sandeels has given rise to the largest single-species annual catch—in some years this has exceeded 1 million tonnes—and constituted more than 40% of the total quantity landed in the North Sea area. In 2000, following concern over the declining productivity of kittiwake breeding, fishing for sandeels in an area off the east coast of Scotland was stopped [2]. However, this management measure was not based on any specific formulated management objectives. A significant decline in sandeel stocks
around Shetland in the late 1980s, coinciding with a marked decline in breeding success of several seabirds, could not be convincingly attributed to industrial fisheries and remains an area of controversy [70]. Studies by the Scottish Office indicated that the decline in sandeel abundance was caused by natural fluctuations in recruitment. Hydrographical changes (temperature) may likewise be an important factor having an impact on sandeel abundance [65]. In the western and north-western parts of the North Sea, there are spatial overlaps between extensive industrial fishing and feeding areas for breeding sea birds. The energetic requirements of seabirds, and the distribution of those birds, are now generally well-known. However, while seabird populations are now monitored all over Europe, key direct responses of seabirds to food shortages, such as the reduced feeding rates of chicks, longer feeding trips, or shifts in diet, are seldom studied on a regular basis [70]. As a consequence, the influence and unexpected side-effects of fluctuations in availability of prey and fishing effort on seabird breeding success is not yet clarified.

There are no explicit management objectives set or any agreed management plan for the sandeel stocks and species in the North Sea [67]. All that has been set for North Sea sandeel are precautionary spawning stock biomass limit reference points, and only general management considerations in relation to the EU precautionary approach have been formulated, including a need to develop management objectives that ensure the stock remains high enough to provide food for a variety of predator species. Also, the management of sandeel fisheries should try to prevent local depletion of sandeel aggregations, particularly in areas where predators congregate [67]. Likewise, only general management objectives are formulated for seabirds through, for example, the EU Conservation of Wild Birds Directive (79/409/EEC).

Ecological Quality Elements for seabirds, identified in the Bergen Convention, include: local sandeel availability to black-legged Kittiwakes and seabird population trends as an index of seabird community health [3]. With respect to the first, ICES [2] recommends that the black-legged kittiwake can be used as an indicator species for the predators that depend on sandeels as an important food source. ICES propose an ECOQO for the first element as follows: Black-legged kittiwake breeding success should exceed (as a three-year running mean) 0.6 chicks per nest per year in each of the following coastal segments: Shetland, north Scotland, east Scotland, and east England [2]. This is considered of high relevance, as it is believed that kittiwakes in the above mentioned areas feed primarily on sandeels. If kittiwakes are unable to breed successfully for a series of years, it is likely that sandeel abundance is low, which, in turn, is likely to have adverse effects on many animal species that prey on sandeels. This reference point is only sensitive when sandeels are at a very low abundance in areas close to bird breeding colonies. With respect to the second ECOQ element, ICES suggests that further investigations of kittiwake population dynamics should be carried out, and that the ECOQO previously suggested by ICES (<20% decline over >20 years) could only act as a precautionary limit to trigger further investigations, but would need to fit into a more advanced framework for ECOQOs before becoming operational. In conclusion, no specific Ecological Quality Objective has been adopted or implemented yet for these ecological elements concerning seabirds [2, 3] and, as such, no specific and restrictive management objectives and limit reference points are in force for them.
Consequently, evaluation of the potential effects on sandeel stock status and dynamics of industrial fisheries, in relation to their effect on local seabird population dynamics and breeding success, will be difficult to achieve before specific management objectives and limit reference points for this relationship are adopted. As a result, it will be difficult to implement an effective operational management plan to manage such effects.

10.3.4 Gillnet and mid-water trawl fisheries in the North Sea—Effects on small cetaceans

Incidental by-catch of cetaceans occurs in several gillnet and mid-water trawl fisheries in Europe, including in the North Sea. The harbour porpoise is the most seriously affected species, as its by-catches seem to be at levels that may greatly reduce some populations. Major declines of harbour porpoise have already been observed in several European coastal areas [88–91].

In some cases, cetaceans actively seek out fishing gear to feed on entangled fish; while, in others, they are not attracted to the gear but encounter it incidentally while feeding or migrating. These different behavioural patterns have implications for the methods of mitigation that might be effective. Most of the behavioural descriptions of these interactions needed to devise and implement optimal, low-energy, mitigation methods are lacking, and, in particular, little is known of the patterns of sonar use in different gear-cetacean interactions [89].

The use of battery-powered ‘pingers’ to frighten away porpoises is, at present, a method of by-catch reduction in gill-net fisheries, which has been shown to work, at least in the short term, although with serious drawbacks [92]. No method, however, exists for reducing mid-water trawl by-catches [89, 93–97]. The employment of high density gillnets also seems to reduce cetacean by-catch, compared to conventional gillnets. This is probably due to the greater stiffness of high density gillnets [92]. In general, the technical properties of fishing gears and specific fishing practices are highly associated with the magnitude of small cetacean by-catch [98].

A vital aspect of managing the by-catch of harbour porpoises is the identification of the spatio-temporal high-risk areas for by-catch in each fishery. However, it should be recognised that price competition and lack of participation in decision-making by fishermen also have significant importance in relation to this problem [89, 98].

General management objectives for the conservation of large whales are formulated through the IWC (International Whaling Commission)—the UN Convention of the Law of the Sea (UNCLOS) states that IWC is the appropriate body to put forward management strategies for these marine mammals. However, there is no such an agreement for small cetaceans, and, in Europe, there is no legislation comparable to the Marine Mammal Protection Act adopted in the USA. Management objectives for the conservation of small cetaceans exist only at the regional level and have been developed through various agreements such as the ASCOBANS (Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas), under the Bonn Convention, [99, 100]. This agreement foresees that adherent countries are obliged to develop methods to reduce by-catch of small cetaceans in the relevant North and Baltic Seas’ fisheries. It has been agreed that the interim objective for harbour porpoise in the North Sea and adjacent areas should be
to restore or maintain species populations at levels equal, at the minimum, to 80% of the carrying capacity; the latter however, has not been defined [100–102]. Additionally, a specific management measure of ASCOBANS (see, for example, [89]) and a management objective set through the ECOQOs (Ecological Quality Objectives) of the Bergen Declaration and ICES [2, 3], is a limit of annual by-catch levels to 1.7% of the estimated stock size. Consequently, through the ECOQOs, specific management objectives concerning small cetacean by-catch have been formulated [2, 73, 74].

The problems faced by small cetaceans in EU waters are also recognised by the EU Commission in the Directive 92/43/EEC (Habitats and Species Directive), where all cetaceans are listed as species in need of strict protection, and where, among others, harbour porpoise is listed as one of the species whose preservation requires the designation of special areas of conservation. Under this Directive, Member States are required to monitor the incidental capture and killing of all cetacean species and ensure that such events do not have a significant impact on the species concerned. The aim of both the Directive and the Bonn Convention is to maintain a favourable conservation status for the species concerned [102]. Even though ratified in ASCOBANS, the EU has not implemented specific management objectives and limit reference points for the conservation and by-catch of small cetaceans in EU fisheries. Recently, the EU has asked its advisory body—the Scientific, Technical and Economic Committee for Fisheries (STECF)—to formulate a management system with specific objectives, management strategies, and harvest control rules and reference points, followed by suggested management measures for small cetaceans based on ASCOBANS [103]. However, the system suggested by STECF has not been yet incorporated in EU fisheries management.

The European Council of Ministers’ Declaration, 812/2004, contains general obligations and management measures for reducing by-catch of small cetaceans in the relevant EU fisheries, without giving specific management objectives and limit reference points. In some cases, national plans have been implemented, such as the Danish Action Plan established in 1998, aiming to reduce porpoise by-catches in the North Sea gillnet fishery. This is mainly through the use of acoustic alarms (pingers) aimed at bycatch mitigation [104].

In conclusion, even though management measures aiming to reduce by-catches of small cetaceans have been applied to North Sea fisheries, no operational management objectives and limit harvest reference points have been adopted or implemented as yet in relation to the management of harbour porpoise by-catch. This makes effective EU management of this issue difficult.

10.3.5 Driftnets for large pelagic species in the Mediterranean—effects on marine mammals

Driftnets are nets that are held on or just below the surface with the help of floats. Their bottom size is properly weighted, so that the tension created between the floats and the weights keep them vertically in the sea. Their height varies, but in the case of large nets, it is around 20–30 m. Nets can drift of their own accord or with the vessel to which one end is tied.

In the Mediterranean, such nets of limited length have traditionally been used to fish a variety of large pelagic commercial species (particularly swordfish and various tunas).
Initially, due to their relatively small mesh size and length, the impact of driftnets on other species was considered negligible. However, problems developed in the 1950s, when the use of synthetic fibres allowed fishermen to increase mesh size and net length (up to 10–12 km) to maximise captures of large individuals. At this point, it became evident that driftnets have a serious impact on several non-target species, particularly marine mammals [105–107].

Typically, marine mammals have a long lifespan, mature late in life, have low reproductive output and rely on a strongly iteroparous reproductive strategy [108]. To offset their low fecundity, they require high rates of sub-adult and adult survival: thus, intense predation by humans on adults or sub-adults is likely to have devastating effects [109].

In the early 1990s, a maximum limit of 2.5 km was imposed on the length of driftnets used by EU fishermen (EC Council Regulation 345/92). However, problems persisted because there were practical difficulties in effectively monitoring this regulation. Finally, a ban on driftnet fishing in the Mediterranean was imposed on the 1st of January 2002 for all EU fleets. Non-EU countries do not enforce this ban, although the United Nations General Assembly has adopted, since 1991, a series of resolutions establishing a worldwide moratorium on driftnet fishing.

### 10.3.6 Bottom trawls in the Mediterranean littoral zone—effects on sensitive habitats

In the Mediterranean, near-shore habitats such as soft substrates and rocky reefs, serve as nursery areas for several marine species. The ecological importance of soft bottoms as nursery areas for different fish species and molluscs has been well documented for several marine areas [110–113]. In the Mediterranean, soft bottoms, which may be sandy or muddy, are sometimes associated with sea-grasses, mostly *Posidonia oceanica*, *Cymodocea nodosa*, and *Zostera noltii*. These sea-grass beds are among the most typical and productive ecosystems in the coastal zone of the Mediterranean Sea, where they occupy extended areas from 0 to 40 m depth [114]. Several authors have documented the importance of such seagrass beds as nursery grounds for a series of marine species in temperate and tropical seas (see [114]).

The shallow zone of the rocky area is characterised by high biotope heterogeneity with alternation of patches of boulders, pebbles, rocky flats, gravel, and sand, with a biotic cover dominated by macrophytes. Many marine species with specific microhabitat requirements settle in this zone [115–117].

Intense anthropogenic activities, prevailing along the Mediterranean coast, have devastating effects on the environment and, particularly, on seagrass beds, which are considered by international conventions to be endangered habitats. *Posidonia oceanica* seabeds, in particular, are experiencing a widespread decline throughout the Mediterranean Sea [118]. In order to reduce the negative effects of fishing on such habitats, the EU has banned trawling within three miles of the coast or in depths of less than 50 m (whatever comes first). This ban, however, is only enforced in the EU waters.
10.4 CURRENT EUROPEAN FISHERIES MANAGEMENT SYSTEMS AND ECOLOGICAL SIDE-EFFECTS OF FISHING

In theory, fisheries management can be approached in two fundamentally different manners: (a) a holistic approach that uses the entire ecosystem as its starting point; and, (b) a single-species approach that focuses only on the species of interest, providing advice which does not take into account the rest of the ecosystem [119].

Holistic approaches are supposed to take into account fundamental aspects of ecosystem structure (for example, trophic interactions). Thus, it is apparent that such approaches represent the “real world” more effectively. Further, as most fisheries are multi-species, it is expected that multi-species management approaches would be most appropriate in the majority of cases [120]. However, due to the complexity of most marine ecosystems, approaches that attempt to model the system as a whole encompass several uncertainties. Apart from the sum of the single species uncertainties, additional uncertainty arises through the poorly understood biological interactions between most of the stocks. Ward [121] identified several “gaps and uncertainties” in the process of deriving marine ecosystem sustainability indicators for Australia’s marine ecosystems. Among others, he identified problems related to: (a) limited ecological knowledge; (b) limited scientific understanding of credible cause-effect environmental issues; (c) the synthesis and aggregation of data; and, (d) implementation issues (such as, case study trials, reference sites and interpretive models).

Although there are tools, such as ECOSIM [122], which allow simulation of dynamic change in a food web with multiple exploited components, balancing exploitation rates between different fleets harvesting different components requires judgements on complex and often conflicting socio-economic issues [123]. Such complexities and uncertainties pose difficulties in defining operational objectives and performance measures.

As a consequence, single-species approaches prevail in the existing management systems and they form the core of management advice. Current European fisheries management depends on stock assessments to estimate population parameters of the focal species from the age or length structure of past catches, biomass of past catches, past fishing effort, and fishery independent surveys [124]. Typically, fisheries scientists formulate potential management actions based on assessment estimates and provide them to managers, who weigh their socio-political consequences in deciding which to implement. In EU waters, one of the most common management methods is the imposition of the total allowable catch (TAC) on fishing for focal species. Other management actions include technical measures and effort limitations. Apparently, existing management systems in Europe are not directly considering ecological bio-indicators or any other ecological side-effects of fishing.

However, at the same time that fisheries management strives to address the shortcomings and uncertainties of single-species approaches, fisheries are increasingly being called upon to consider the ecosystem perspective. In particular, fisheries are forced to consider international conventions and agreements dealing with biological diversity, ecologically sustainable development, conservation of the environment, and the protection of endangered and threatened species. Nature conservation issues arise as a result of the localised effects caused by fishing as well as cumulative impacts that result at the ecosystem
level. Such concerns have both given rise to a growing body of research and have also contributed towards international and national agreements, conventions and directives giving general management objectives for conserving biodiversity and putting users of the environment (for example, fisheries) on an ecologically sustainable footing. These are, for example: the Convention on Biological Diversity; UNCLOS; UN World Summit on Sustainable Development; UN Convention on Straddling Fish Stocks and Highly Migratory Species; Reykjavik Declaration on Responsible Fisheries in the Marine Ecosystem (2001); Declaration from the World Summit on Sustainable Development (Johannesburg 2002); FAO Code of Conduct; OSPAR Convention on the Protection and Conservation of the Ecosystems and Biological Diversity of the Maritime Area; ASCOBANS; the EU Common Fisheries Policy (EU Green Paper); EU Council Habitats and Species (92/43/EEC) and Birds (79/409/EEC) Directives; The Bergen Declaration [2, 3]; and Nordic Council of Ministers Agreements [125]. Such initiatives have increased the pressure for change from a broad variety of stakeholders and directed attention when nature conservation issues arise [126].

The “precautionary approach” to fisheries management, as outlined by the FAO Code of Conduct for Responsible Fisheries, recognises that fishing can have unpredictable and irreversible effects on marine ecosystems [127]. FAO compiled 137 paragraphs describing the features of the “precautionary approach” in relation to fisheries, and Restrepo et al. [128] provide the following concise definition: “In fisheries, the precautionary approach is about applying judicious and responsible fisheries management practices, based on sound scientific research and analysis, proactively (to avoid or reverse overexploitation) rather than reactively (once all doubt has been removed and the resource is severely overexploited), to ensure the sustainability of fishery resources and associated ecosystems for the benefit of future as well as current generations”.

In the EU context the Bergen Declaration [2, 3] has attempted to set some specific management objectives and limit reference points for a small number of side-effects of fisheries: for example, for the spawning stock biomass of commercial fish species; and for the by-catch of harbour porpoises. However, specific management objectives and limit reference points are missing for many important ecological side-effects of important EU fisheries.

The way that single-species approaches consider environmental aspects is by modifying single-species management targets, or other aspects of the management strategy, based on qualitative or semi-quantitative analysis of species interactions, habitat effects, and effects on non-target species. For example, a single-species quota may be set according to some benchmark, but fishing may be restricted further in certain areas or seasons to avoid local depletion of fish species consumed by protected or endangered species, to reduce by-catch of non-target species, or to minimise impacts on fragile bottom types [119]. The examples mentioned in the previous section illustrate some management actions that have been employed in line with the above approach.

Management tools that are used to consider the negative effects of fishing on the ecosystem include technical measures and control effort regimes. Technical measures focus on improvements to fishing gears and minimum landing size regulations. Fishing gear improvements refer to a range of technological solutions aiming to improve selectivity. Apart from simple increases in the mesh size of nets to reduce juvenile catches, which
have been extensively applied in various fisheries, other more sophisticated devices have been developed. These include acoustic “pingers” that can be attached to set nets to alert cetaceans to their presence, bird-scaring devices to deter seabirds from attacking baited long-lines, and by-catch reduction devices, such as panels and grids, to prevent catches of marine mammals and sea-turtles in trawling gears. [89, 95–97].

The control of fishing effort is a basic tool of fisheries management and fishing effort is usually treated as a continuous variable that can be controlled indirectly through quotas or even by controls on fleet tonnage or vessel/gear characteristics. An examination of fleet replacement strategy, however, shows that new vessels commonly enter a fishery as ‘pulses’ of new capacity whose fishing intensity is then difficult or impossible to control by indirect measures [129]. Although seasonal and long-term area closures were among the earliest tools to be used in fishery regulation [130], and continue to be an important method of protecting nursery grounds and sensitive habitats in certain areas (e.g., Mediterranean), area closures seem also to have gained in popularity as a result of the evident difficulties in predicting fishery impacts on the ecosystem [129].

Spatial closures amount to the creation of “marine protected areas” MPAs. The definition of MPAs is very broad, but based on the diversity of practical uses and ecological settings that MPAs include, the World Conservation Union (IUCN) has classified them in six categories [131]. As outlined by Harmelin et al. [21], the establishment of MPAs can be beneficial in providing refuge to threatened species, protecting ecosystems or habitats with particularly high biodiversity, facilitating recovery of already damaged areas, protecting breeding stocks, improving recruitment to neighbouring areas, or restocking marine species of commercial interest. MPAs in which all harvesting activities are prohibited, also called “marine reserves”, have the potential to reduce overall fishing mortality, to preserve species diversity and to protect ecosystem integrity [132]. They can also provide information on the effects of human disturbance on fundamental processes at the population, community and ecosystem levels [133].

The use of MPAs as a tool to meet a broad array of management objectives is of increasing interest. Although further studies are needed to clarify the mechanisms operating in MPAs, there is growing evidence that area closures benefit a wide range of marine species with different life history characteristics [132, 134, 135]. As there is an increasing public concern regarding the effects of fishing on the environment, and fisheries management systems are being forced to consider the ecosystem perspective more actively, the use of MPAs as a management tool will probably become a more central issue.

10.5 CONCLUSIONS

Due to the complex nature of the marine ecosystem and the lack of appropriate ecosystem monitoring schemes (these being resource demanding), our knowledge on the impacts of fishing on non-exploited species and the marine environment as a whole, is rather limited and fragmented. Natural variations and impacts of other human activities (both marine and terrestrial) on the marine environment may also mask the effects of fishing.
Consequently, only a limited understanding of the impacts of fishing on non-target species and on the marine environment in a more holistic perspective has been achieved.

Within current management systems, the adverse effects of fishing on the ecosystem are only partially addressed, mostly through realisation of general objectives stated in the EU Green Paper, the FAO Precautionary Approach to Fisheries, and in various international conventions and agreements. Progress in achieving specific and operational management objectives that address the interactions between fishing, exploited species, and the wider ecosystem has been slow, even in systems under single national jurisdictions.

Instead of only managing fisheries to avoid the extinction of certain species or destruction of certain exposed and sensitive marine habitats, management objectives should reflect optimal management in relation to the ecologically sustainable exploitation of marine resources. Clear and operational management objectives are critical to the successful development of an ecosystem approach to management. As different countries and a variety of different stakeholders, with very different backgrounds and understandings of the system, will have different perceptions of management targets and support different approaches to attain and meet objectives, consensus on those objectives between groups will be the major challenge for the implementation of an ecosystem approach within fisheries management [1].

Even though Ecological Quality Objectives are in the process of being developed within OSPAR, the Bergen Convention and the ICES framework, and being evaluated by EU, the definition of ecosystem reference points and key indicators is at a very early stage. Further work on this aspect is necessary to successfully evaluate and manage the ecological side-effects of fishing and establish an ecological approach to fisheries management.

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Ecological side-effects

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Chapter 11

Fisheries-Based Management and Advice in Europe

Wim Demaré


11.1 INTRODUCTION

The management of fisheries in European community waters is controlled by the European Union. Every year in December, the Council of Ministers—made up of the Ministers for Fisheries from the different Member States—fix the rules for the fishing opportunities of the coming year. Usually, the negotiated Council Regulations for fishery management are based on a proposal from the European Commission. The Commission is split into administrative departments or Directorates General (DGs), of which DG XIV is in charge of fisheries. DG XIV relies on national and international scientific research for its proposal to the Council. For example, the International Council for the Exploration of the Sea (ICES) assembles information on more than 100 North Atlantic fish and shellfish stocks, which is used to give the Commission advice on fisheries management. Besides, de Regional Advisory Councils (RACs)—made up of professional fishing organizations and non-governmental organizations—can give recommendations to the Commission.

The management of fish stocks that are shared with non-EU Members is organized via regional fisheries bodies and/or agreements. There are for example several stocks in the North Sea (Cod, haddock, whiting, plaice, herring, etc.) that are shared between Norway and the European Union. The total allowable catches for these stocks are set with bilateral agreements between the two parties. There are also shared Mediterranean stocks between the European Union and some African countries. Management measures for these Mediterranean stocks are arranged via the General Fisheries Commission for the Mediterranean (GFCM). A similar situation applies to the Baltic, where management measures are settled via the International Baltic Sea Fisheries Commission (IBSFC). The European Union is also a member of the North East Atlantic Fisheries Commission (NEAFC)—dealing with fish stocks in the Atlantic and Arctic Oceans, of the Northwest Atlantic Fisheries Organization (NAFO)—to manage fisheries outside the Extended Economic Zones, and of the International Commission for the Conservation of Atlantic Tuna (ICCAT).

Fishery-based advice and management are currently ‘hot items’, especially in the North East Atlantic region of Europe, where traditionally stock-based advice and management is found. This section deals with the presence of fishery-based measures in the current
EU legislation and with the effort made to include the fishery-based approach in the European advisory and management system. Special attention is given to the European Common Fisheries Policy (CFP), the most important instrument of the European Union to manage its fisheries.

11.2 CONTEXT OF THE EUROPEAN COMMON FISHERIES POLICY

On 1 January 2003, the new European Common Fisheries Policy (CFP) entered into force. Its main objectives are that the CFP “shall ensure exploitation of living aquatic resources that provides sustainable economic, environmental and social conditions” [1]. These goals have not, fundamentally, changed since the implementation of the CFP in 1983 [2], but it has now become apparent that the CFP has not always been successful in achieving them. Most stocks are not exploited sustainably and employment in the fishing community is declining, frequently because of poor financial profitability. Many possible explanations for this failure have been identified. These include: that decisions have been taken from a short-term, rather than long-term, perspective; that Commission proposals have been overruled by the Council of Ministers; and that the European fleet suffers from overcapacity.

The effectiveness of the Total Allowable Catch (TAC) management tool, one of the cornerstones of the CFP in the North East Atlantic (NEA) region, can also be questioned. Most demersal fisheries—such as the flatfish-directed beam trawl fisheries—in this region are mixed. And for these fisheries management by TACs is less appropriate, especially when stocks are in need of preservation, than they are in single-species fisheries—such as some pelagic fisheries. This is because, in mixed fisheries, it is almost impossible to target one species without catching other species, for which fishermen may not have quota. This can give rise to logbook irregularities, such as underreporting and misreporting of the landings; and to a high rate of discarding. The latter occurs for two key reasons relating to the TAC system: first, fishermen are not allowed to land fish for which they do not have quota; and, second, quota scarcity incentives high-grading of the catch, which is fish that have already been caught are discarded in favour of a higher quality and more valuable catch within the same fishing trip. The TAC system thus frustrates fishermen who are faced daily with the problem of how to handle surplus catch, the administrative people who have to enforce and control the legislation, and the scientists who have to work with inaccurate landing data. The current TAC management in the context of a mixed fishery also makes it very difficult to conserve one species in the mixed fishery without affecting the profitability of the whole fishery. Thus, there is a need to take account of the mixed nature of some fisheries and to evolve from stock-based to fishery-based management and advice.

The Mediterranean fisheries are mainly managed by technical measures, which are often fishery-based. However, these have not been a success either, since most demersal and pelagic resources in the Mediterranean have been considered overexploited. In view of this, the measures should be re-evaluated in order to reduce the mortality on juveniles and to reduce the overall fishing effort of those fisheries catching the overexploited stocks [3, 4].
11.3 FISHERY-BASED MANAGEMENT AND ADVICE IN THE CFP

Since the establishment of the CFP, TACs and technical measures have been key-tools for the conservation and management of fisheries resources in Europe. In general TACs\(^1\) are stock-related while technical measures are often fishery-related. Technical measures include (1) spatial and/or temporal closures, (2) measures regarding fishing gear and its method of use, and (3) minimum landing sizes (MLS). In the previous and current term of the CFP more fishery-based measures were explicitly added to the list of measures: more selective fishing measures and measures to reduce the impact of fisheries on marine ecosystems and non-target species\(^2\).

11.3.1 The North East Atlantic (NEA)

For both the Mediterranean and the NEA, technical measures are laid down in key legislation\(^3\). This legislation operates within a long-term management policy for the conservation of fish resources. In the NEA, this long-term strategy is supplemented with short-term measures—mostly TACs but also additional technical measures with respect to certain fisheries. For example for the recovery of Irish Sea cod, a temporal and spatial closure is in place for most Irish Sea demersal fisheries, and flatfish-directed beam trawlers are obliged to fish with a panel of diamond-meshed netting material with mesh size greater than 180 mm.

TACs are widely used for the management of stocks in the NEA. While TACs might be appropriate for single-species fisheries, they are not ideal for mixed fisheries. Although introduced originally to address national allocations of commercial fish resources within the European 'common pond', one of their purposes has been to indirectly regulate effort. However, in practice, effort is difficult to control via TACs. Other drawbacks of the system are that it encourages misreporting and discarding. Therefore, a changeover from a stock-based TAC system to a more fishery-based advice and management regime is the next logical step. The first moves in that direction occurred in 2003. Because of the poor status of some cod stocks, strong TAC reductions for these stocks were deemed necessary. Since cod is regularly caught in mixed demersal fisheries, this measure could only be effective if TACs from associated stocks were also decreased. Consequently, haddock and whiting TACs were further reduced at that time. This approach—which is mixed species, rather than pure fishery-based—is now applied to all recovery stocks in the NEA and tries to take the mixed nature of some fisheries into account. More recently, fisheries on recovery stocks have been subject to effort restrictions: for example, the cod recovery plan dictates that most demersal fishing operations can only operate for a

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\(^1\) For single species fisheries TACs can in theory be considered to be both stock-based and fishery-based measures. Consequently, TACs are under much less fire in single species fisheries than in mixed species fisheries. For the remainder of the text, TACs are mainly discussed in relation to mixed species fisheries.

\(^2\) The full text of all CFP regulations can be found in Council Regulations [1, 2, 17].

\(^3\) The key technical legislation is given in Council Regulations [18, 19] for the NEA (except for the Baltic) and in Council Regulation [20] for the Mediterranean. Technical measures for the Baltic are laid down in the “Fishery rules of the International Baltic Sea Fishery Commission”.
limited number of days in the North Sea. Until now, the fishery-directed approach has been focused mainly on recovery stocks. However, it will be expanded to all NEA stocks caught in mixed fisheries. After all, the most recent Memorandum of Understanding (MOU) between the International Council for the Exploration of the Sea (ICES) and the European Community states that “for each sea area, ICES shall define groups of stocks within which ICES shall ensure close quantitative consistency between advice given for each stock. This should be considered a first step in the development of fisheries-based advice”. ICES is the scientific organisation that provides the European Commission with information and advice on the status and exploitation rates of over 100 North Atlantic fish and shellfish stocks. Traditionally, this advice is given on a single-stock basis. As a result, it might well occur that fishing mortality for one stock is allowed to increase, while fishing mortality for an associated stock must decrease. Under these circumstances, it is obvious that this will cause problems when put into practice as a fisherman fishing on both stocks simultaneously, must increase his fishing pressure on one stock, but must at the same time also decrease his fishing pressure on the associated stock. To avoid these conflicting signals, which are typical of a single-stock advisory system, a fishery-based approach is being gradually integrated into the ICES advice. A more fishery-based approach to the example described above would be that the associated stock, for which the fishing pressure has to decrease, determines the advice for the other stock. This makes the advice consistent between stocks. Albeit that this consistency is not quantified at the moment, ICES has begun to give qualitative information on inter-stock and fisheries relationships. Such relationships can vary from high to low intensity, depending on whether most fisheries catch the stocks together or only a few of them do. For example, Celtic Sea cod, haddock and whiting are highly linked in some trawl fisheries. The same cod stock is, to a lesser extent, taken together, among other stocks, with anglerfish, hake, megrim, three sole and plaice stocks, rays and a couple of Nephrops stock. Each of these stocks can be linked to each other or to other stocks, which complicates the quantification of the consistency between advice for different stocks and explains the difficulty of integrating such inter-relationships into the advice.

11.3.2 The Mediterranean

In the Mediterranean, fisheries can be categorised into (1) fisheries targeting highly migratory fish, (2) fisheries targeting shared demersal and small pelagic stocks or fisheries operating in shared areas, and (3) fisheries targeting stocks primarily distributed in national waters and caught by one Member State only. Fisheries of the first two categories are expanding and are to be managed at an international and European Community level, fisheries of the third category are to be managed at national level [3].

At the European Community level, fisheries are subject to a technical regulation, often supplemented with local legislation or agreements by fishermen. Despite these stock conservation measures (mostly fishery-based), the Commission concluded in its Green paper [4]:

Until now the technical measures Regulation for the Mediterranean has not been a success. There may be a need to seriously re-evaluate mesh sizes and landing sizes. There is also
a need to consider the introduction of an effort-control management scheme in the absence of TACs. The current inability of the General Fisheries Commission of the Mediterranean (GFCM) to adopt such a scheme for the whole area should make the Community reflect on the initiatives that need to be taken on her part. (pp11)

Upon this, a Community Action Plan for the Mediterranean Sea was communicated in 2002 [3] and a number of actions were considered to achieve the objectives of the CFP in the Mediterranean. It was decided that the basic management tools should be fishery-based measures to improve selectivity and to regulate fishing effort rather than single-species approaches and/or output measures. This philosophy was confirmed in November 2003 at the Ministerial Conference for the Sustainable Development of Fisheries in the Mediterranean in Venice. At this conference, the development of resource management and conservation mechanisms, based in particular on rules on fishing effort and technical measures, including Fisheries Protection Zones (FPZs) and improvement of gear selectivity, was recommended. Although the application of the effort approach is considered to be the most important management instrument for the Mediterranean, it has also been suggested that TACs may still be suitable [3]. Currently, bluefin tuna is the only Mediterranean stock subject to a TAC. However, according to the Commission, stocks such as Mediterranean swordfish, albacore and some small pelagic stocks, are candidates for future TAC management if they are fished in clearly targeted fisheries where they dominate the catch composition. This ties in with the view that TACs operate optimally in a single-species context, rather than in a mixed fishery.

An additional complication for the Mediterranean is that both European and African fleets fish on some stocks. In 2004, the trans-Mediterranean association of fisheries organisations, Medisamak, was founded, because of the need for improved international co-operation between European and African stakeholders in the search for sustainable fisheries. International management of the shared stocks is put aside for GFCM. This Commission was established under an international agreement under the aegis of the FAO. Their purpose is to promote development, conservation, rational management and the best utilisation of living marine resources of the Mediterranean and the Black Seas. For the management of Mediterranean stocks and fisheries, GFCM focuses on fishing effort as a management tool, rather than on TAC applications as these are considered to be inappropriate for controlling mixed species fisheries.

11.4 TOWARDS FISHERY-BASED ADVICE AND MANAGEMENT

11.4.1 The MTAC model

The mixed-species TAC evaluation or MTAC model [5, 6] is a computer programme that aims to translate the mixed nature of some fisheries into appropriate management measures. The model was first used some years ago to address the mixed nature of fisheries taking cod in the NEA, as ICES recommendation, at that time, was for a cod moratorium. Since most demersal fisheries catch cod, this would imply inactivity for a major part of the demersal fleet. Such a management decision would be unacceptable to
the fishing industry for obvious social and economic reasons. On the other hand, within this context, if the demersal fleet kept on fishing at the same rate, cod mortality could never be reduced. A compromise was the only solution. Thus the key question was: how could the demersal fleet still operate without harming the cod stock too much? The level of allowable catches for stocks closely related to cod is an important issue here.

The MTAC model can make these calculations. The model gives fishing mortality multipliers per stock, translated into TACs. MTAC is used as a kind of TAC reallocation procedure. In that sense, the MTAC approach is mixed species-based rather than fishery-based. It attempts to resolve the conflict—due to the mixed species nature of the fisheries—between the traditional single species catch forecasts [7].

The current format of the data going into the model is mainly fleet-based, rather than fishery-based. To clarify this, concepts like fishing activity, métier and operational unit must be introduced. Although these terms differ somewhat in meaning, the concept behind these definitions is fundamentally the same. They create a framework for data collection, and for advice and management. The terminology of these concepts is given in the Report of the Ad Hoc Meeting of Independent Experts on Fleet-Fishery-based Sampling [8] and is summarised below.

(i) Fishing activity. A group of fishing trips targeting the same species, using similar gear, during the same period of the year and/or within the same area. Appropriate aggregations of fishing activity types are (sometimes) the basis for biological sampling in the NEA. Fishing activity is referred to as fishery or métier [9].

(ii) Fleet. A group of fishing vessels sharing, during a reference period (e.g., one year), similar characteristics in terms of technical features, economic structure and/or major activity. Vessels may have different fishing activities during the reference period, but they can be classified in only one fleet. The fleet is the unit for sampling economic data (and sometimes biological data) in the ICES area.

(iii) Operational unit. Group of fishing vessels practicing the same type of fishing operation, targeting the same species or group of species and having a similar economic structure. The grouping of vessels may be subject to change over time, and depends on the management objectives to be reached [10]. During a year, a vessel may switch between operational units. However, a vessel belongs to the same fleet segment during that year. The operational unit combines the concepts of fleet and of fishing activity. The operational unit is meant to be the basis for sampling both biological and economic data in the Mediterranean.

So far fleet-based data has been fed to the MTAC model, allowing, in theory, the calculation and allocation of fishing possibilities per fleet. The programme calculates fleet-factors (fishing mortality multipliers per fleet), which could be used for a more fishery-based advisory and management approach. However, these factors are not currently used for this purpose, because there is, as yet, no political basis on which to do so (especially in relation to the principle of relative stability: see below). Nor would it be straightforward to implement them. In the North Sea for example, more than 70 fleets have been identified and it is impossible to implement a management framework, based on quota, for each
Consequently the MTAC model is probably not going to solve the mixed fishery problem. However, it effectively illustrates the complexity of it.

11.4.2 Data


The European Member States have to submit an annual National Data Collection Programme, according to the rules stipulated in the DCR, and have to assure the collection of (i) data on fishing capacity, (ii) data related to fishing effort, (iii) data related to catches and landings, (iv) data concerning catches per unit of effort, (v) survey data, (vi) length and age samples from landings and discards, (vii) other biological parameters such as maturity and sex ratio, (viii) economic data by group of vessels, and (ix) data concerning the processing industry.

The DCR has resulted in a better and more harmonized collection of data on fisheries and fishery related issues in the European Union. Before the DCR, the data collection did depend on the initiative taken by the individual Member States, whereas at present the different countries are obliged to take their responsibility with regard to data gathering. Much is expected from the discard sampling programmes stimulated by the DCR and set up for the first time on a regular basis in many Member States. Discard estimates are indispensable for the fisheries advisory and management process (especially for fisheries with high discard rates). A major drawback of the DCR is that the spirit has been stock-based rather than fishery-based, so that in some cases a Member State is obliged to sample a stock, but not a fishery. This makes for example no sense in on-board sampling. In that case the observer will sample the catch of the commercial vessel, and thus the fishery will be sampled. In addition, fishery-based advice and management can only be achieved if the data are collected accordingly. This criticism has been taken up by the Commission and will be accounted for in the next revision of the DCR. The way forward is nicely summarized in the Report of the Second Regional Co-ordination Meeting for the North Sea [13]. “Independent experts met in Nantes on behalf of the European Commission to report on fleet/fishery based sampling [8]. During this meeting, there was agreement to base the future collection of data for fisheries in a matrix-like format, splitting the information by group of vessels and by métier/fishery. The purpose of this approach being to: (i) propose a common frame for economists and biologists and to (ii) define more accurate stratification for sampling and international coordination”.

11.4.3 Relative stability and alternative management regimes

In the NEA, the absence of a political will to change the principle of relative stability is a major impediment to switching from stock-related to purely fishery-based management. Relative stability of fishing activities, by the allocation of fishing opportunities among
the European Member States, based upon a predictable share of the stocks, is one of the basic principles of the CFP. In other words, each Member State gets a quota or fixed share of the TAC. The quota are defined in weight and are directly related to each individual stock. For that reason a sheer fishery-based approach in mixed fisheries and the current stock-associated relative stability principle contradict one another, for a fishery of a certain Member State might get a quota for one stock, but not for another associated stock. Purely fishery-related management demands an abolition or redefinition of the principle of relative stability. For the latter, the stability should be based on the concept of fishery (taking into account the dynamic environment of that concept), and not on a weight percentage unit as is now the case.

Effort management must be considered as a fishery-associated measure. Regulating fisheries via the input (effort), rather than via the output (TAC), is a potential way forward for implementing fisheries-based management. Both systems have their advantages and disadvantages. A major drawback of the TAC application is the misreporting and discarding. Fish can be allocated to adjacent areas, black landed or thrown back to postpone quota-exhaustion or when quota are fished out. When quota are becoming exhausted, fishermen can select fish sizes that have the most value, a phenomenon called highgrading. Under an effort regime, these practices should diminish. Black landings to avoid taxes and highgrading-will nevertheless occur under all management schemes. With an effort system, misallocation of effort becomes possible, although it should be easy to control with satellite monitoring. There is also a risk that vessels will concentrate on the more ‘attractive’ species. This can be favourable when attractive means abundant, but it can also work out badly when stocks—even when they are in a critical state—remain attractive because of their economic value.

Recently, days at sea restrictions have been imposed on fisheries fishing on so-called recovery stocks. Thus the management of mixed fisheries in Europe is evolving from a stock-based TAC management to a TAC management complemented by fishery-based effort limitations. A changeover from a TAC to an effort-only regime in Europe is, however, unlikely and would generate the problem of how to allocate effort fairly, taking into account the currently defined principle of relative stability. Shepherd [14] proposed to apply all changes of effort pro rata to all those affected. This means that the fishing effort of every participant in a fishery or métier should be adjusted up or down by the same proportion. For example if beam-trawler fleets operating in the Celtic Sea have to reduce their effort by 10%, every single beam trawler can spend 10% less time fishing in the Celtic Sea. Initial effort should be based on historic records of activity. Shepherd [14] argues that this approach implements the principle of relative stability and, at the same time, avoids the need to decide on the equivalence between different vessels and gears. Yet, as fisheries are a dynamic system, rules should be set up to exchange effort between vessels of similar fisheries, or on how vessels can switch between fisheries. With such rules and under such a regime, the currently defined relative stability principle has no raison d’être.

A noteworthy problem that has to be accounted for in an effort regime is increasing fishing efficiency or technological creep. Although effort adjustments could and should be made for increased fishing efficiency, the Icelandic example shows that, when the demersal fisheries are managed by effort alone, both fleet size and fishing effort (by technology
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creeping) increased [15]. At the beginning of the Icelandic effort regime, trawlers were allowed to fish for 323 days; four years later, the allowable fishing days were 215 days. When the system changed to individual transferable quota (ITQ), the size of the fishing fleet was slowly reduced and demersal fishing effort declined substantially [15]. The ITQ system has brought the Icelandic demersal fishery towards greater efficiency and the success of the system is related to the creation of private property in harvesting rights. In Europe, the Netherlands partly apply an ITQ scheme to their TAC share, but an expansion to other Member States may result in taking more advantage of the ITQ system4. If Europe would implement an ITQ regime (for which the principle of relative stability might be a starting point to put the system into practice) problems like discarding and misreporting would probably remain, but experience from abroad has shown that this approach is much more efficient in controlling effort. As for the TAC system, an ITQ regime can at the most be mixed-species-based rather than sheer fishery-based.

CONCLUSION

Two different approaches towards fisheries management are in operation in the NEA and in the Mediterranean, but stocks from both areas suffer from overexploitation. In the Mediterranean, most fisheries are managed through technical measures, which are often fishery-based. The poor status of most stocks urges more stringent actions in this area. TAC management is not the preferred option, but could be applicable to target stocks when they comprise the majority of the catch composition. Involved parties are now focusing on developing more effective rules, in particular based on fishing effort and technical measures such as Fisheries Protection Zones and gear selectivity improvements.

In the NEA, TACs are the main management tool and these are intrinsically stock-based. To take the mixed nature of some fisheries into account, TACs of associated stocks should be more geared to one another. This is also defined in the MOU between ICES and the European Community. Defining the linkage between associated stocks can become very complex, especially in multi-species and multi-fisheries context. Even if these linkages could be uncovered and quantified and even if one could develop a system of quota allocation per fishery, the implementation of such a regime would be hardly manageable. The management of mixed fisheries in Europe is evolving from a stock-based TAC regime to a management of TACs, which accounts for mixed species interrelationships and that is complemented by fishery-based effort limitations. It is unlikely that the management of fisheries in Europe will change to an effort-only regime. Besides, the implementation of such a regime would be severely hampered by the current principle of relative stability. Whatever the management regime is, it should be able to effectively control effort in order to harvest stocks sustainably. Fishing effort in Europe is currently much too high in relation to fishing opportunities, with the Commission recommending reductions in effort of approximately 40%, and in many cases much higher [16].

4 For an extensive description of the ITQ regime see Chapter 3 on ‘Right-based fisheries management’.
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Chapter 12

The Requirements of an Ecosystem Approach to Fisheries Management

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12.1 INTRODUCTION

Over the last decades the emphasis of fisheries management has changed from optimizing production to conservation and risk management [1]. During the same period the knowledge about the impacts of fishing on marine ecosystems has been vastly improved. It is now well-known that fishing has impacted seabed habitats, by-catch species and overall ecosystem structure and function in many parts of world. The trophic level of global fish landings has declined [2], and studies have linked fishing to changes in the size structure of exploited fish assemblages [3], thus altering trophic interactions and pathways of energy flow. In some areas reductions in the abundance of large predatory fish species have been substantial [4] and recovery of depleted fish populations has been reported to be slow [5]. Size selective exploitation has led to changes in the genetic makeup of target fish populations [6], and by-catch mortality has impacted non-target fish populations, benthos, seabirds, marine mammals and turtles [7, 8]. Active fishing gears in contact with the seabed have affected benthic assemblages and habitats [9] and discarding has provided large quantities of food to scavenging species [10].

The increasing public awareness of environmental issues has led to the recognition that marine ecosystems not only provide marketable fish and shellfish, but also a whole range of other ‘goods and services’ to humans, and research has documented that some of these good and services may be at risk at current levels of fishing effort. As reflected by many recent policy documents and declarations, summarized in FAO [11], this has created a motivation for broadening fisheries management to include objectives related to the wider impacts of fishing on marine ecosystems.

At the international level the motivation can be traced back to the 1972 UN Conference on Human Environment and the 1982 UN Convention of the Law of the Sea. Since then the concepts and ideas behind an Ecosystem Approach to Fisheries Management (EAFM) have gradually evolved in parallel with improvements in the understanding of the impacts of fishing on marine ecosystems and the need to protect and conserve them. The UN Conference on the Environment and Development (1992) defined sustainable development as ‘meeting the needs of the present generation without compromising the ability of future generations to meet theirs’, and introduced the concept of precautionary
management. In Agenda 21, it was emphasized that protection of marine ecosystems and use of marine resources were inseparable, and that “new approaches to marine and coastal management and development [that are] integrated in content and are precautionary and anticipatory in ambit” were needed. The Convention on Biological Diversity (1992) called for conservation of biodiversity at the genetic, species and ecosystem levels. According to the convention the right to exploit biological resources entails an obligation to manage activities that may threaten biodiversity. The Reykjavik Declaration on Responsible Fisheries in the Marine Ecosystem (2001) specifically addressed the issue of introducing ecosystem considerations in fisheries management. It recognized the complex interrelationships between fisheries and marine ecosystems, and called for immediate introduction of management plans with incentives for sustainable ecosystem use. In the declaration from the World Summit on Sustainable Development in Johannesburg (2002) the heads of states agreed to protect and restore the integrity of the planet’s ecological system, with special emphasis on protecting biodiversity. In the implementation plan it was agreed to “encourage the application by 2010 of the ecosystem approach, noting the Reykjavik Declaration on Responsible Fisheries in the Marine Environment”, to “maintain productivity and biodiversity of important and vulnerable marine and coastal areas”, and to “develop and facilitate the use of diverse approaches and tools, including the ecosystem approach, the elimination of destructive practices, the establishment of marine protected areas . . . and the integration of marine and coastal areas into key sectors”.

The increasing emphasis on sustainability and nature conservation in international agreements and conventions has been matched by a large number of scientific publications and symposia on the issue. The symposium on the Ecosystem Effects of Fishing [12], the scientific sessions of the Reykjavik Conference on Responsible Fishing in the Marine Ecosystem [13] and the conference on Quantitative Ecosystem Indicators for Fisheries Management [14] has summarized a lot of the work and progress in understanding of how fishing affects marine ecosystems and what an ecosystem approach to management entails.

12.2 THE ECOSYSTEM APPROACH TO FISHERIES MANAGEMENT—WHAT IS IT?

12.2.1 The concept

The concept of an Ecosystem Approach to Fishery Management (EAFM), or as other label it, Ecosystem-based Fisheries Management, is rapidly evolving as evidenced by the increasing number of publications on the subject, providing a large range of opinions regarding its definition and how it may be achieved.

In the FAO Guidelines on the Ecosystem Approach to Fisheries [11] EAFM is broadly defined: ‘An Ecosystem Approach to Fisheries strives to balance diverse societal objectives, by taking account of the knowledge and uncertainties of biotic, abiotic, and human components of ecosystems and their interactions and applying an integrated approach to fisheries within ecological meaningful boundaries’ and the purpose of the approach was
described as ‘to plan, develop, and manage fisheries in a manner that addresses the multiple needs and desires of societies, without jeopardizing the options for future generations to benefit from the full range of goods and services provided by marine ecosystems’.

Including ecosystems considerations in fisheries management is intended to broaden traditional fisheries management aimed at securing the sustainable exploitation of the target high-value species, by addressing ecosystem concerns such as impacts of fishing on by-catch species, changes in prey availability for predator population and the effects of fishing on habitats [15]. According to Sissenwine and Murawski [16] the EAFM “is geographically specified, takes account of ecosystem knowledge and uncertainties, considers multiple external influences, and strives to balance diverse societal objectives”. It includes considering by-catch and discarding, habitat impacts, indirect effects of harvesting and interactions between biological and physical components of ecosystems in fisheries management plans.

For many the EAFM is seen as a more inclusive approach than traditional fisheries management, but not as radical or revolutionary departure from traditional fisheries management concepts. Some believe that EAFM will require a change of current fisheries management institutions, because it necessitates integration of policies across diverse sectors. Still others see EAFM as an attempt to solve just one of the problems of current fisheries management systems, but consider the major problem to be the lack of appropriate incentives and tenure systems to break the race for fish [17]. Garcia and Cochrane [18] summarized the current situation and provided a comparison between the properties considered in traditional single species fisheries management and EAFM. Although the scope of EAFM is much wider than conventional fisheries management its practical implementation has similar components including the setting of overall high-level objectives at the policy level together with the identification of responsible institutions, user rights and mechanisms for conflict resolution and decision. To guide management action the overall high-level objectives must be translated into operational objectives reflecting the issues of concern e.g. by identifying acceptable levels of fishing impacts on target species, non-target species and habitats. The formulation of the management plan for a given fishery includes the identification of the stakeholders, a description of the fishery and the area in which it takes place, the identification of the broader issues, the specification and prioritization of the objectives, an agreement on indicators and reference points, identification of management measures and decision rules, and plans for monitoring, assessment and review.

Broadening the scope of fisheries management from considering the sustainable exploitation of the target species to include ecosystem and conservation objectives requires improved mechanisms for dealing with multiple objectives and trade-offs [19]. Most fisheries management systems have been designed to maximize single species landings within relatively simple conservation limits, and it was not envisaged that they would be required to deal with complex interactions between biological components or user groups and to consider objectives related to nature conservation alongside with objectives related to sustainable exploitation [20]. Changing management institutions to enable them to deal with increased uncertainty and complexity, multiple stakeholders and integration of policies across different sectors will be one of the main challenges of EAFM.
12.2.2 Management objectives

Many of the policy documents describing EAFM refer to high level objectives such as “maintaining the structure and function of marine ecosystems including their biodiversity”. The environmental NGO’s perspective is that the conservation of biodiversity, or prevention of irreversible impacts to marine ecosystems should take precedence over maximizing fisheries yields, arguing that the loss of biodiversity, whether at the genetic, species, or ecosystem level, represent a huge loss of opportunity costs in the future [21], while the fishing industry itself often weight short term catch opportunities much higher than longer term issues. Ethical and value-based standpoints play a strong role in the argumentation between what seems to be a continuum of viewpoints on marine conservation. At one of this continuum some argue that nature has a right in its own and hence should be preserved [22], while the other, utilitarian point of view, regards humans (fishermen) as an integral part of marine ecosystems, and view nature conservation as an essential component of management plans aimed at securing their sustainable exploitation.

By studying the wider impacts of fishing on marine ecosystems natural scientists have played an important part in the specification of potential overall objectives for EAFM. According to Larkin [23] there are three broad goals in the ecosystem approach. It should secure a sustainable yield of products for human consumption and animal foods; maintenance of biodiversity; and ensure protection from the effects of pollution and habitat degradation. Gislason et al. [12] emphasized maintenance of ecosystem and species diversity; genetic variability within species; sustainable exploitation of directly affected species and; minimization of impacts on ecologically dependent species and overall trophic composition, while Pikitch et al. [24] considered that the overall objective of ecosystem-based fisheries management is to sustain healthy marine ecosystems and inferred that this could be achieved by avoiding degradation of ecosystems, as measured by indicators of environmental quality and system status; by minimizing the risk of irreversible change to natural assemblages of species and ecosystem processes; by ensuring that long-term socioeconomic benefits can be obtained and maintained without compromising the ecosystem; and by generating knowledge of ecosystem processes sufficient to understand the likely consequences of human actions, and where such knowledge is inadequate, by adopting robust and precautionary fisheries management measures that favour the ecosystem.

While natural scientists have played a major role in identifying the components of EAFM, specifying its overall objectives is clearly not just about science [25]. For the ecologist and other natural science disciplines involved in natural resource management, the discussion of what constitutes desirable and undesirable ecological states of nature cannot be taken in isolation from discussions within stakeholder groups driven by social, economic and political issues and agendas. This creates a need for providing a structured framework where the communication between scientists, NGOs, fishers, the industry and other stakeholders can take place and in many areas such frameworks are being generated. An increase in the transparency of the management system to all stakeholders is an obvious necessity, for a successful communications, but much more could be done to identify the most efficient way to achieve consensus and to identify and develop methods and institutions to further the dialogue.
12.2.3 Making high level objectives operational

Once overall objectives have been defined they have to be translated to operational objectives related to measurable properties. Overall objectives are intended to reflect broader policy commitments, and are often not be sufficiently specific to guide management action. If the overall objective for example is to “maintain the structure and function of marine ecosystems including their biodiversity”, it will be necessary to consider a number of ecosystem characteristics such as trophic relationships, species and genetic diversity and habitat quality, as well as the long term status of affected populations and their resilience to natural and man-made environmental changes. All of these characteristics can be expected to be influenced by fishing. For each ecosystem characteristic that fishing may influence operational objectives have to be agreed upon in order to identify what constitutes acceptable and unacceptable impacts.

In this context it can be useful to distinguish between direct and indirect impacts of fishing. Direct impacts are those which are directly observable and linked to fishing such as the landings of target and by-catch species removed extracted from the ecosystem, the amounts discarded and the size of the areas impacted by physical disturbance of the seabed. In the longer term these impacts intermingle with natural changes caused by species interactions and natural or man-made environmental changes and fluctuations leading to longer term changes in population sizes, energy fluxes as well as in the genetic make-up of affected populations. Direct impacts are relatively easy to quantify, while the quantification of the longer term indirect impacts requires sustained monitoring of relevant variables, multivariate analysis and modelling to allow the effects of fishing to be separated from natural fluctuations and global environmental change. The scientific literature is rich with examples of documented or likely effects. Lists of potential effects to consider are given in the guidelines developed by FAO [11] and in numerous publications elsewhere [12, 13], and quantitative indicators to measure these effect are being developed [14].

12.2.4 Indicators

An indicator can be defined as a quantifiable variable, pointer or index that relate to a criterion [26]. In EAFM indicators are typically needed to reflect the state of the ecosystem, the fishery resource and the economic and social state of the industry in relation to operational objectives. Indicators are useful to inform managers, stakeholders and the public about how well the objectives are being pursued and met and determine what properties should be monitored. For each operational objective one or several indicators will have to be selected to track how the situation develops. Typically an objective will be represented either by a preferred value of the indicator or by a limit value reflecting the border between acceptable and unacceptable states, but it can also just be whether or not the indicator is moving in a desired direction.

A number of criteria are useful for identifying appropriate indicators [27–29]. These include how simple and concrete the indicator is, ensuring that decisions are easy to explain to stakeholders and the public; the validity of its theoretical and scientific basis; the public awareness of the property it is supposed to reflect; the costs, uncertainty and
potential bias involved in its measurement; the availability of historical data to provide a baseline for evaluating change; and the sensitivity of the indicator to changes in fishing and other management actions including the timescale of its response and whether it may respond to other drivers, e.g., environmental forcing.

The selection of appropriate indicators is not a trivial process. Ideally it involves an iterative process in which candidate indicators are screened by stakeholders for their usefulness in relation to management objectives, while scientists advice on the ability of the indicators to reflect the impact of fishing on the property of interest and the ease with which each indicator can be monitored. The involvement of stakeholders in the screening of indicators is important, because indicators must be legitimate and meaningful in the eyes of the stakeholders and the public. Indicators proposed by science are sometimes based on abstract models, and such indicators may not be legitimate or meaningful to stakeholders if these have little or no academic training. Degnbol [30] found a large discrepancy between research-based knowledge and stakeholder knowledge on the effects of fishing on target species and ecosystems and concluded that the identification of a common ground where objectives could be shared should involve not only longer term sustainability issues, but also issues relating to short term resource allocation as these were often of over-riding importance for the stakeholders. This is in line with Garcia and Cochrane [18] who noted that even though traditional fisheries management pretend to aim at conservation of exploited resources, in reality the main concern is conservation of livelihoods and employment.

A large number of indicators have already been proposed [14]. While the virtues and drawbacks of single species indicators and reference points related to spawning stock abundance or fishing mortality have been well-described in the fisheries literature, indicators reflecting aggregate system properties and socioeconomic performance are still under development. With respect to aggregate system properties Rice [31] identified five classes of possible indicators: indicator species; diversity indices reflecting species richness and how similar their abundances are; ordination methods summarizing data on changes in species abundances into one or a few major axes; aggregated indicators such as size spectrum slope or dominance curves, and; metrics of emergent properties such as resilience to perturbations. The conference on Quantitative Ecosystem Indicators for Fisheries Management provides examples of additional indicator categories reflecting the spatial and trophodynamic organization of marine ecosystems [14]. Defining operational goals and selecting appropriate indicators to characterize the state of the ecosystem is hampered by the limited knowledge about how ecosystems respond to fishing, and how the effects of fishing can be separated from the effects of natural environmental change. Many of the ecosystem indicators proposed will thus be summarizing the outcome of a large number of processes of which only few will be understood in detail [1]. Daan et al. [3] remarked that few of the more general ecosystem indicators have yet been underpinned by scientific documentation of a clear quantitative relationship between the response of the indicator and the pressure generated by fishing, and this obviously limits their applicability. This has led some to suggest that the most tractable solution in the short term would be to expand the single species reference points and performance measures used in traditional single species fisheries management to also cover non-target species, ecologically dependent species and species affected by scavengers [29]. To this list could
be added indicators reflecting the genetic selection differentials caused by size selective fishing. It is likely that many of the impacts of concern can be dealt with pragmatically by reducing effort to sustainable levels and minimizing adverse side effects of different gear types and fisheries, leaving only a subset of concerns relating to impacts on overall ecosystem properties.

Due to the complexity of marine ecosystems and the multitude of fishing activities and stakeholder aspirations a large number of indicators will probably be needed to accurately reflect the situation in relation to a range of diverse management objectives. However, using a large number of indicators will diminish the ability to overview the situation and methods are therefore being developed to aggregate indicators to provide an interpretable overall picture. This can either be done graphically by use of score cards or kite diagrams [32] or by using multivariate methods to extract the overriding signal from a variety of indicators e.g. [33]. However, indicators have to be accepted by stakeholders as legitimate, and using multivariate methods to group indicators with similar patterns of co-variation may result in composite indicators with unclear interpretation and little direct linkage between effect and cause thus disguising features important for decision-making [28].

Another possibility is to construct conceptual frameworks to group the indicators. García and Staples [26] reviewed a number of frameworks for classifying indicators. The Pressure–State–Response (PSR) framework was developed by OECD and other international bodies. In a fisheries context pressure indicators would include fishing effort of various gear types, while state indicators could include proportion of exploited fish stocks below their limit reference biomass; average size of fish, overall trophic composition, habitat impacts etc.; and response indicators could be the success of effort reduction measures, the proportion of total area protected from fishing, by-catch reduction measures etc. In order to improve its reflection of the social, economic and institutional dimensions of sustainable development the PSR framework has been expanded into the DPSR framework, where D stands for driving forces such as economic and demographic processes explaining the origin of the pressures, but it is still a fairly simple framework because it ignores many of the linkages and feedbacks between the components. Changes in the average size and abundance of fish could thus be expected to change the economic performance of various fisheries and this could feed back into changes in effort. Chesson et al. [34] have described the Australian framework for sustainable development of fisheries known as the Ecologically Sustainable Development (ESD) framework in which a hierarchy of items to consider. The primary subdivision in the framework is the dichotomy between the effects of fishing on humans and the effects on the environment, where the latter is divided into target species, non-target species and other aspects (e.g., habitat effects), and the effects on humans are categorized as food, employment, income and lifestyle. Such frameworks can ease the overall selection of indicators and guide discussions on their relative importance.

In many cases emphasis has been on identifying indicators relevant for communicating information to decision-makers, and less has been vested in the role of indicator frameworks for developing management policy experiments. Rudd [35] proposed to use a modified version of the Institutional Analysis and Development (IAD) framework of Ostrom [36] to organize indicators and test theories and models for linking institutions
and sustainability in common pool resource systems. The IAD encompasses both the process-oriented PSR and the structurally oriented sustainable livelihoods frameworks and provides possibilities for considering multiple ecological and socioeconomic indicators in a common framework.

12.2.5 Dealing with uncertainty, complexity and subtle change

Uncertainty plays an important role in EAFM. With far more conflicting objectives to consider than in single species management and with possible effects ranging from changes in single species productivity to changes in ecosystem energy flux, biodiversity and species genetics, it is evident that the likelihood of an effect occurring will have to be considered together with its potential severity. Quantification of uncertainty and risk will therefore play an important role in the road towards EAFM.

The complexity of marine ecosystems and the lack of knowledge to link observed changes at the ecosystem level directly to fishing pressure makes it difficult to identify appropriate indicators. It is evident that many aspects of how fishing affects marine ecosystems still remain unknown or highly uncertain [37]. In many cases little information was collected before a fishery developed. Although palaeoecological and historical studies can provide some information [38] too little data is often available to characterize the situation before exploitation began. Without knowing the past, it can be difficult to evaluate the present, and fisheries biologists and marine ecologists run the danger of using a situation already influenced by fishing as their reference for evaluating further change [39]. Furthermore the recent evidence for global climate change will soon make it difficult to use data from the past to extrapolate the future, as environmental conditions will no longer be comparable.

Classical ecosystem models may offer a tool to link pressure to state, but these models should be used with caution, due to the limited understanding of the underlying processes that govern ecosystem structure and function and the problems involved in parameter estimation. Robinson and Frid [40] reviewed the usefulness of existing marine ecosystem models for evaluating ecosystem effects of fishing and although 8 out of 24 models provided a reasonable coverage of most of the functional groups relevant for an assessment, none covered all of the possible impacts and very few dealt with socio-economic issues. In the absence of models covering all aspects of concern some have proposed to use methods such as Fuzzy Cognitive Maps to translate expert knowledge into predictions of the effects of management on ecosystems [41].

Although significant progress has been made in modelling the properties of marine ecosystems there are still significant unresolved issues. There is first of all substantial uncertainty about the appropriate level of complexity of the models and the functional form of the relationships included [42]. Second, the models often require large amounts of data and parameter estimation is difficult. Different sources of input data will have to be weighted according to their variance, but it is often non-trivial to model the statistical properties of the data, and hence to specify appropriate likelihood functions, due to the over-dispersion caused by patchiness and correlation in time and space [43]. This makes it very difficult to test ecosystem models with sufficient statistical rigour. Sufficiently
long time series of relevant ecosystem properties, funds to collect the necessary additional data, and expertise to validate the models will only be available in a few areas of the world [1].

In addition to the uncertainty about how fishing changes overall ecosystem properties, the implementation of EAFM faces a number of other problems [25]. As Sinclair and Valdimarsson [13] point out “a first step in moving towards ecosystem-based fishery management is to identify and describe the different ecosystems and their boundaries, and then to consider each as a discrete entity for the purposes of management. Thereafter, ecosystem management objectives must be developed.” However, most marine ecosystems are difficult to delineate and even where this can be done a mismatch between the spatial extension of ecosystems and the jurisdiction of various management authorities is often observed [12]. Ecosystem boundaries are difficult to define and rarely match the boundaries of the constituent species and populations. Adjacent ecosystems will typically exchange nutrients and biota. Many species perform large scale migrations and pelagic eggs and larvae may drift over large areas. There can be considerable genetic differences between subpopulations and stock components, the significance of which is poorly known. Management problems must obviously be dealt with at local regional, national and international levels. This can foster international and regional cooperation, but result in no-action being taken, because national or regional initiatives will be undermined by actions of those in other jurisdictions and thus anyhow be ineffective. Given the diversity of scales involved a system of nested institutional arrangements seems to be the best solution, but an effective design of such a system necessitates an understanding about how actions at one level change the incentives of the actors at other levels [44].

Second, questions pertaining to marine environmental protection, shipping, extraction of oil and gas, mariculture and fisheries are often dealt with by different management authorities representing different industries and interests, and these authorities sometimes have conflicting policies. Cross-sectoral coordination, if not a re-arrangement of authority, is needed to ensure consistent management. EU-policies have thus been criticized for their efforts on one hand to reduce fishing effort by vessel decommissioning schemes, and on the other to encourage the building of new capacity [45].

Third, there is a mismatch between the time frame in which policy makers are elected and the time it takes for significant improvements to occur. In the eyes of the fishing industry fisheries management frequently involves balancing short-term loss against uncertain long-term benefits. Evidence from stock collapses, such as northern cod, and empirical studies of the response of fish communities in no-take marine reserves has shown that the response time can be considerable [5]. Northern cod has not yet rebounded and Russ and Alcala [46] found response times of 15 and 40 years, respectively, for large predatory reef fish in two marine reserves in the Philippines. The future is often discounted relative to the present and it is therefore necessary to extend the time horizon of managers to secure that sustainability and intergenerational equity is preserved [19]. The power of current monitoring programs to detect significant changes is furthermore limited [47, 48] and it will often be necessary to regulate fishing long before a change can be proven to be statistically significant.
It is important to recognize the trade-off between knowledge and risk. Lack of knowledge entails higher risk and calls for precautionary management and more stringent measures. Increased knowledge, on the other hand, is expected to lead to lower risk provided the management system is able to utilize the knowledge efficiently. Where considerations of risk become part of the management procedure there is no need to stick to the 5% significance level of classical statistics. The level of acceptable risk should be open for discussion among stakeholders and management authorities representing society at large, and not be fixed by scientific default.

In a situation with high uncertainty and insufficient funds to monitor the situation, the obvious option is to reduce the catch sufficiently to make it unlikely that adverse situations arise. There is, however, scope for improving the cost effectiveness of fisheries management to reduce uncertainty without increasing costs. Some of these possibilities include adopting new technology to monitor the catch and by-catch (video and image analysis), location and activity (satellite based vessel monitoring) and reporting (electronic logbooks); using the industry to collect information; improving catch control rules to minimize risk; reducing by-catch and environmental impact by changes in gear design, and establishment of MPAs to safeguard a proportion of the ecosystem. Important lessons may also be learned from other industries facing similar problems with uncertainty and risk. Commercial enterprises generally have to deal with uncertainties regarding supply, demand, inflation and exchange rates, and have found methods to cope with these. However, in fisheries management the trade-off between monitoring costs and catch opportunities are often not made explicit to the industry. Using taxes and license fees to pay for monitoring costs and fleet reduction programs may encourage the industry to take a greater responsibility [49].

EAFM must be able to cope both with the uncertainty associated with predicting the change in goods and services produced by complex ecosystems and with the organizational and institutional complexity of integrating management across at various spatial, temporal, and sectoral scales. It must also be able to accommodate the expected increase in ecological understanding and improvement of models of marine ecosystems with time. Management goals and strategies will have to evolve as knowledge and understanding improve. This is not going to be unproblematic. Climate is likely to change due to global warming and indicators could respond in unexpected ways to changes in ocean environment, as this will change pathways of energy flow and biodiversity at different trophic levels as well as the distribution, recruitment and production of major commercial fish stocks. A consistent change in climate will make it increasingly difficult to apply past experiences as a guide to management and poses severe constraints on the use of adaptive management, where the outcomes of deliberate changes in management are used to guide the development of future policies. It is important that work is undertaken to study when and how reference points should be updated to account for climate change. Using past conditions as a reference will probably be increasingly difficult to defend and management institutions must be able to adapt to changing environmental conditions and changes in knowledge. Orstom [44] concludes, that when it comes to governance of natural resources the most successful institutions are those that are able to modify their rules over time according to a set of collective and constitutional choice rules.
12.3 MANAGEMENT TOOLS

The management tools available for EAFM are generally the same as used in single species management, but three tools can be considered to be of particular relevance. Marine protected areas are relevant because they can be designed to protect vulnerable habitats, provide a refuge for target and non-target species, and add robustness to uncertainty about fishing impacts and effectiveness of other management measures. Gear modifications can be used to avoid habitat damage and reduce by-catch and discarding of unwanted species and sizes. Eco-labeling will assist by creating incentives for environmentally friendly and sustainable fishing practices and by increasing consumer influence on how fisheries are conducted and managed.

12.3.1 MPAs

Many consider Marine Protected Areas to be an important management tool in the context of EAFM [50–52].

Established MPAs range from strictly “no take” areas to “multiple-use” areas where some types of fisheries and other activities are allowed. Often they were created to simultaneously fulfil several objectives related both to fishery and non-fishery purposes, such as tourism and habitat conservation. The multitude of MPA objectives and their order of priority vary therefore from place to place. They have become popular in fisheries management because they have the potential to offer protection to vulnerable habitats, species and critical life stages; reduce overall fishing mortality by providing refuges for target populations and by-catch species and lead to increases of spawning stocks and potentially to enhanced recruitment; provide increased catch opportunities to fisheries just outside their borders by the spill-over of individuals; mitigate the selective effects of fishing on species genetics by providing an area free of fishing impacts; and provide a buffer to management failure in the surrounding areas.

Particularly for small scale fisheries in developing countries, where limited institutional capacity for enforcement and monitoring make traditional catch control and effort reduction measures unlikely to work, MPAs have been seen as the only feasible solution to the management problem. However, MPAs provide only limited protection for migratory species and their establishment may displace fishing effort to nearby areas, where it can lead to increases in fishing mortality, by-catches, habitat disturbance and discarding. For this reason few would agree that they can solve all fisheries management problems and most concur that they should be regarded as an important supplement to other management measures.

Many existing MPAs have been designed on an ad hoc basis without the necessary ecological and socio-economic information and planning available to ensure that their objectives could be met. As a result only 31% of MPAs was reported to meet their management goals in the survey made by [53]. Experience has shown that MPA proposals often generate conflicts between users who differ in their views on conservation and exploitation [54]. To many fishermen MPA proposals are not acceptable in areas where they fish, unless they can be convinced that it will lead to increased catches in adjacent areas or provide alternative income, e.g., through ecotourism [55]. Unfortunately the
benefits derived from their establishment in terms of increases in tourism within and catch opportunities outside their borders have often not been able to substitute or exceed the value of the landings lost. The resulting lack of local community support has, in combination with poor enforcement, often led to continuation of fishing inside their borders. In some areas it has therefore been decided to create incentives for conservation by granting territorial user rights to local users by allowing them to exclude others, thus securing that the benefits of conservation and sustainable exploitation accrue to those who bear the costs [56]. One of the major problems is the no-use value of ecosystem integrity and biodiversity. The long term value of biodiversity is potentially very large, but in many countries this has limited relevance for policy unless compensation mechanisms are developed to transfer income to the local stakeholders who bear the costs of conservation.

Another problem with the design of present MPAs is that their sizes often have been so small that it was unlikely that they could reach their stated ecological objectives. Many marine species have a pelagic larval stage allowing them to disperse over large areas. If MPAs are placed where currents does not allow a sufficient number of recruits to settle within their borders, the expected population increase inside the MPAs will never emerge. While MPAs may protect sedentary species once they have settled, they cannot be expected to offer similar protection to species characterised by highly dispersive life-stages. To increase their efficiency some MPA proponents argue that at least 20% of an each habitat type should be protected from fishing as a default [57, 58] and networks of MPAs are now recommended, where ecological knowledge and modelling is used to decide the size of the individual MPAs, their placement and how they should be connected [59, 60].

Inshore areas are often characterized by multiple uses, with different social sectors valuating different aspects of coastal marine ecosystems. This often creates conflicts. Even where activities are spatially separated, the connectivity of marine ecosystems means that impacts may spread across the borders of MPAs. Their management must therefore often address a range of conflicts originating from activities outside their borders [54]. Educating stakeholders on the social, economic, cultural, and ecological benefits of MPAs and allowing them to participate is likely to increase their success. Likewise planning efforts benefit if scientists and managers are educated on the social, economic, and political implications of establishing MPAs and their effects on stakeholder livelihoods [61].

The evidence for benefits of marine protected areas has been summarized in a large number of reviews, so large in fact that the rate of publication of reviews and theoretical papers recently seems to have exceeded the rate of publication of empirical results [62]. Most reports provide evidence of increases in biomass and abundance of species of commercial interest within no-take areas. Reports documenting benefits for adjacent fisheries outside their borders are rarer. Murawski et al. [63] found significant increases in catch per unit of effort for some commercial species close to the borders of large-scale closed areas off northeast USA, and increases in effort along the borders, and concluded that this was due to a spill-over of fish from inside the areas. Similar increases in catch was found by Roberts et al. [64] outside closures of 35% of the fishing area on the coral reefs of St. Lucia.

Sale et al. [65] summarized the major gaps in the biological knowledge necessary for designing networks of MPAs. From a biological point of view the most needed information
concerns the connectivity and anticipated recruitment to adjacent areas. Sale et al. [64] noted a lack of sufficient knowledge about dispersal of larvae and movement of adults determining rates of self recruitment to the protected area and the potential spill-over of adults to adjacent areas; limited ability to predict changes in trophic interactions and competition within the areas; insufficient knowledge to predict the passive transport of larvae and juveniles between adjacent reserves and thus to characterise localities as either sources or sinks of propagules; and remarkably few well designed studies to demonstrate that a MPA had sustained or enhanced the fishery in the adjacent region. In addition to this there are a number of deficiencies in the understanding of socioeconomic and social issues at the sites where MPA are placed. Research is needed into the proper combination of MPAs with other management tools to achieve a given set of objectives. Pomeroy et al. [66] used a set of evaluation guidelines to compare the management of 18 MPAs throughout the world. Together with managers, planners, and other decision-makers a total of 42 indicators of socioeconomic conditions, governance structure, ecological status and human impact were selected and used for to assess the effectiveness of the MPAs with respect to 21 management objectives. While the results are preliminary they demonstrate that such comparisons may provide guidance to improve the management performance of MPAs.

Overall MPAs have an important role to play in the establishment of EAFM. Although the benefits for local fisheries are doubtful, well-planned MPAs may safeguard vulnerable habitats, populations and genetic resources. Supplemented with measures to manage fishing outside their borders and established with a reasonable size they seem able to provide a number of the conceivable objectives of EAFM.

12.3.2 Gear modifications

Although estimates vary from fishery to fishery, by-catches often constitute a significant part of the catch brought on deck. By-catch regulations play an important part in the management of several fisheries and new by-catch reducing devices are continuously being developed and tested. Considerable efforts have in recent years been used to make fishing gears more selective, minimize their side-effects on benthic habitats and fauna and reduce the by-catch of unwanted species and size groups.

In trawl and seine fisheries increases in mesh size or shape have contributed to reduce the by-catch of small sized fish. Sorting grids and escape windows have been able to reduce the by-catch of larger non-target organisms in fisheries for small bodied species such as shrimp [67]. By-catches of turtles in shrimp fisheries has for example caused considerable public concern, and this has led to the development of Turtle Exclusion Devices, which are now mandatory in many trawl-fisheries for shrimp throughout the world. Other gear modifications, such as separators or raised footrope designs, have reduced the by-catch of unwanted species of fish and other biota.

Towed gears in close contact with the seabed will disturb the sediment, dislocate stones and boulders, and dislocate or injure benthic animals. Beamtrawls, dredges and the boards of ottertrawls will dig into the sediment and leave detectable marks. Beamtrawls are supplied with tickler chains designed to scare flatfish and shrimps up from the seabed and into the net, while dredges either scrape the surface for epifauna such as scallops
or penetrate the seabed to harvest clams and other infauna. There are no easy ways to reduce the adverse environmental impacts of these gears, but attempts are being made to develop alternative less disturbing methods, such as electrical stimuli, for scaring flatfish up from the seabed.

In purse seine fisheries for tuna in the eastern Pacific Ocean the incidental annual capture of several hundred thousand dolphins in the 1960s led to the development of a series of changes in gear design and operating practices. Among these were the insertion of a section of small meshed netting in the part of the purse seine most often in contact with the dolphins to keep them from becoming entangled; the ‘backdown’ procedure, where the vessel is put in reverse to make the corkline of the purse seine sink thus creating an escape path for the dolphins; the use of rescue rafts and other means of hand rescuing the dolphins from the net; and the training of the fishers on using these methods and other ways of avoiding dolphin by-catch [68]. This reduced the by-catch to very low levels. In so-called dolphin sets where the fishermen use the presence of dolphins as a sign of tuna, approximately 40% of the sets had zero dolphin mortality in 1986; by 1996 this proportion had increased to about 88%. During the same period the average mortality of dolphins per set decreased from over 12 individuals to 0.33.

Gillnets may entangle birds, turtles and marine mammals. Seabird by-catch can be reduced by strips of highly visible netting in the upper part of the net and acoustic pingers can be used to reduce the by-catch of small cetaceans. Baited longlines may catch seabirds trying to eat the bait near the surface during setting [69]. This can be avoided by using bird-scaring lines with plastic strips above the longline or by setting the line through a tube making the bait inaccessible to the birds. By-catches in trap fishing can also be reduced by careful selection of mesh sizes and design of entrances and escape openings. Both gill nets and traps may continue fishing after being lost, but use of biodegradable material in their construction can reduce the period in which they continue fishing.

Although technical improvements in gear design and more precise navigation can decrease or remove some of the negative side-effects of certain fishing gears and practices it is unlikely that all impacts can be reduced to acceptable levels. Often the improvements come with a cost to the industry; will reduce the efficiency of the gear towards marketable species or sizes; or make the gear more difficult to operate. In situations where the economic efficiency of the gear is reduced fishers are likely to modify the gear to increase its efficiency. This may neutralize the anticipated environmental benefits. Only if economic or other incentives are combined with education and proper surveillance will more environmentally friendly gears be rapidly adopted and used by the industry. It is fortunate that many of the technologies developed in recent years to reduce by-catch of non-target species and size groups can be adopted without major reductions in their long term profitability [67]. The reduction in dolphin by-catch in tuna fisheries and the introduction of turtle exclusion devices in shrimp fisheries demonstrate that significant improvement can be achieved if the sufficient public attention and political will is present.

12.3.3 Eco-labelling

Fisheries management has been successful in developing market and tax incentives for fleets to expand, but has generally failed to develop incentives to promote sustainability
Ecosystem approach to fisheries management

and nature conservation, e.g., by supporting by-catch reductions or encouraging the use of gear types with limited environmental side effects Hanna [20]. If fisheries management is to become EAFM it is important to identify incentives that will promote sustainability and nature conservation. Eco-labelling constitutes a market based incentive, where consumers are given an opportunity to express a preference for seafood caught by well-managed and environment friendly fisheries through their selection of labelled products. Consumer preferences for eco-labelled products can help to align the goals of the industry with those of management agencies and environmental NGOs.

Certification may provide a market advantage and certification costs will therefore often have to be paid by the industry. This result in a shift in the burden of proof from management agencies towards the industry, providing the basis for a better understanding in the industry for the need for adopting precautionary and environmentally friendly management systems [70]. The threat of losing certification will create incentives for incorporating appropriate levels of precaution in the management strategy.

Eco-labelling has been used to reduce by-catch of dolphins in tuna fisheries by labelling of so-called “dolphin safe” tuna products and in the marine aquarium trade with ornamental fish [71]. However, the most elaborate program for eco-labelling of fisheries and fisheries products has been developed by the Marine Stewardship Council (MSC). Currently approximately 3% of the global landing from marine fisheries have been or are in the process of being certified by MSC, including large scale demersal fisheries such as the north Pacific Pollock fishery. MSC was established by WWF and UNILEVER in 1999 and has since then developed into an independent, global, non-profit organisation [72]. The MSC definition of sustainable fishing builds on the FAO code of Conduct for Sustainable Fishing which has been further elaborated through an international consultative process with various fisheries stakeholders. According to the definition a well-managed and sustainable fishery is characterized by the following principles:

- A fishery must be conducted in a manner that does not lead to overfishing or depletion of exploited populations and, for those populations that are depleted, the fishery must be conducted in a manner that demonstrably leads to its recovery.
- Fishing operations should allow for the maintenance of the structure, productivity, function and diversity of the ecosystem (including habitat and associated dependent and ecologically related species) on which the fishery depends.
- The fishery should be subject to an effective management system that respects local, national and international laws and standards and incorporates institutional and operational frameworks that require use of the resource to be responsible and sustainable.

These overall principles are further elaborated into a large number of criteria described in the manual used by the certifiers in their evaluation of a given fishery [73].

The certification scheme established by the Marine Stewardship Council is voluntary. The actual certification is undertaken by one of several independent certifiers approved by MSC. A fishery wishing to obtain the certificate will have to hire an approved certifier to do the evaluation. Typically, the process will begin with a confidential pre-assessment of the likelihood that the fishery can eventually pass the criteria and be certified. If the fishery on this basis decides to continue with a full assessment the certifier will collect
the necessary information and make a draft assessment where performance indicators related to the criteria will be scored in relation to operational reference points specific to the fishery being evaluated. Stakeholders and the public will be consulted and provided opportunities to provide input and comments to the selected indicators and reference points, comment on the draft assessment report, and participate in the selection of experts to perform a peer review of the report. Depending on the outcome of the peer review the certifier will recommend whether a certificate should be issued. After the review MSC will collate the final report including the decision, the peer review and the stakeholder comments. The report will be made publicly available on the MSC web site. Stakeholders will then be given the possibility to submit formal objections to the decision and these objections will be considered by an objection panel appointed by MSC.

Since a large number of indicators may be used to score the fishery in relation to the criteria, and since each of these indicators may take different values, special software is used to calculate an average scoring for each of the three overriding principles. In order to receive the certificate the average score should be above the minimum level considered to be in accordance with the MSC standard for each of the three principles. In addition none of the indicators can be below the level where the situation is considered unsatisfactory. If an indicator is above the level considered unsatisfactory, but not yet at the level considered in accordance with the MSC standard, the certifier will specify mandatory actions to improve the performance within the certification period.

The certificate is valid for a period of five years. However, annual and random surveillance audits of the performance of the fishery will be undertaken during the certification period and if performance declines or if the implementation of the actions specified in the final assessment report does not progress at a reasonable speed, the fishery may loose the certificate before the five-year period has ended. To retain the certificate after the end of the five year certification period a new full evaluation will have to be performed. To ensure that labelled fish products originate from certified fisheries and are not mixed with products from any other fishery during processing or distribution the MSC can also issue chain of custody certificates.

The procedures adopted by MSC are designed to recognize that management efforts are most likely to be successful when there is full co-operation among the entire range of stakeholders, including those who depend on fishing for their food and livelihood. However, full agreement is sometimes unachievable and MSC has been criticized by environmental NGOs for not requiring sufficient evidence to evaluate whether adverse ecosystem impacts take place. In relation to the ecosystem objectives mentioned under principle two the fishery is scored according to three criteria requiring that:

1. The fishery is conducted in a way that maintains natural functional relationships among species and should not lead to trophic cascades or ecosystem state changes.
2. The fishery is conducted in a manner that does not threaten biological diversity at the genetic, species or population levels and avoids or minimizes mortality of, or injuries to endangered, threatened or protected species.
3. Where exploited populations are depleted, the fishery will be executed such that recovery and rebuilding is allowed to occur to a specified level within specified time
frames, consistent with the precautionary approach and considering the ability of the population to produce long-term potential yields.

Sutton [74] criticized the decision to certify the Western Australian Rock Lobster fishery, arguing that current knowledge was insufficient for assessing the performance of the fishery in relation to its possible adverse impacts on biodiversity and on the ecological role of the lobsters in the ecosystems in which they live. Instead of claiming that the certificate signifies that no adverse ecological impacts are generated by the fishery he proposed that MSC changed the criteria to emphasize intent, e.g., by using “The fishery is conducted in a way that aims to maintain ...” rather than “The fishery is conducted in a way that maintains ...” in the formulation of the three criteria.

It is unknown to what an extent eco-labelling will improve the general sustainability of capture fisheries. The experience with eco-labelling is still rather limited, but as the number of fisheries seeking certification is growing, this will probably change. Teisl et al. [75] examined the dolphin-tuna controversy and found that the subsequent implementation of dolphin-safe labelling affected consumer behavior in a way that increased the market share of dolphin-safe labelled canned tuna.

12.4 CONCLUSIONS

The Ecosystem Approach to Fisheries Management (EAFM) is in its infancy and fisheries management institutions worldwide are struggling to operationalize the concept and understand its implications on current practices and policies. Marine ecosystems are complex and it is difficult to separate the effects of fishing from the effects of natural changes in environmental and biological drivers. While direct effects can be observed and quantified, the ability to predict the longer term effects of fishing on marine ecosystems is limited. This generates ample room for debate between stakeholders and calls for improved mechanisms for conflict resolution and handling of uncertainty.

The successfulness of EAFM depends on all relevant stakeholders being heard; on efficient means for communicating information about the uncertainty involved in the assessment and the risks involved in different management plans; on proper identification of relevant stakeholders, on comprehensive consultations about objectives, indicators and possible reference points and performance measures; on improved knowledge about how to separate the impact of fishing from the impacts of natural environmental change and other anthropogenic activities; and on the selection of appropriate management measures and incentives to make policies work [29]. It is important that the current uncertainty about the longer term and indirect effects of fishing on marine ecosystems does not stall efforts to establish EAFM. Many of the likely direct impacts of various fishing activities are known and even in data-poor situations it should be possible to use general ecological principles and meta-analysis of results obtained elsewhere to develop precautionary measures such as effort limitations, area closures, and gear improvements to increase the likelihood that the overall impact is sustainable [24].

Using science to create knowledge interactively by inviting stakeholders to define the topics of particular interest and concern, participate in the collection and evaluation of the
scientific evidence, and deciding on the credibility of the results, could change the role of fisheries science in the management system. Eco-labelling will provide stakeholders with a possibility to participate in the discussion of what constitutes valid knowledge, improvements in gear design to reduce the most obvious direct impacts of fishing on habitats and non-target organisms may improve the public perception of the industry, and results from monitoring and analysis of the response of biota within carefully designed MPAs, may make the impacts of fishing much more discernable to the public. In many years fisheries management has been a battleground between management authorities, the industry and fisheries research institutions. Broadening the objectives and involving additional stakeholders may end up creating a new and more transparent role for science, where the translation of scientific knowledge into management measures becomes a more democratic and participatory process and less of a technocratic and expert driven exercise.

Involving local stakeholders will generate local solutions. Ecosystems and stakeholder preferences differ, and different operational objectives will definitely emerge in different parts of the world. There is a need for developing default standards for target and limit reference points to guide local management to live up to national and international expectations. While agreement on high level overall objectives seems to be achievable, experience show that stakeholders often disagree on the actual measures to reach the objectives. Natural science can link cause and effect and help to quantify uncertainty and risk. Social science is needed to create governance structures to further agreement on management measures despite uncertainty and conflicting operational objectives, and to invent socioeconomic incentives to help the objectives being achieved.

REFERENCES

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Chapter 13

Delivering Complex Scientific Advice to Multiple Stakeholders

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13.1 INTRODUCTION

A feature that distinguishes natural resource based industries from other industries is their need for management. The lack of explicit ownership over these resources generally leads to their overexploitation unless some form of intervention is undertaken to regulate and, in some cases, allocate resource use. For most resource-based industries, some form of property rights based system has been introduced or evolved that reduces the degree to which intervention is required. For example, most agricultural land is in private ownership, while access to individual forest and mineral resources has generally been limited to a small (often singular) number of exploiters.

While rights-based management has been applied to many fisheries, these rights are often less well-defined compared with the terrestrial resource based industries. In particular, the fisheries stocks are an unseen, fugitive resource. That is, the size of the resource is not known with certainty as it cannot be directly observed, and the resource moves making both the determination of the state of the resource and the exploitation of the resource more complex. Further, and unlike most other natural resource-based industries, rights of access to the resources have generally been open until (historically) recently. As a result, the resources are perceived as ‘different’ to other resources (which have a long history of limited or exclusive access), and there is a general resistance by many traditional fishing communities to measures that limit their access. Introducing such measures is made more complex by the fact that information on the state of the resource is highly uncertain, while the short-term, direct effects of limiting access (either in terms of vessel numbers, activity or catch) on the local communities is highly certain. Policy makers are often reluctant to implement measures that may have short-term negative consequences on fishing communities if the long-term benefits are uncertain.1

1 A number of hypotheses have been proposed as to the reluctance to adopt rights based systems in fisheries. Bromley [57] argues that the form that rights take is a function of the ratio between the value of a unit area of a natural resource and the costs of enforcing those rights. Open access will be found where the unit value is
The long-term question is how to develop ongoing management that is able to adapt to changing ecological, social and economic circumstances. The benefits of participatory systems includes the ability to incorporate localised knowledge into the decision making process as well as improved compliance through the perception of greater legitimacy of the decisions. Globally, co-management has a strong track record in improving fisheries management [1]. Indeed, it is hard to imagine how a management system could be truly adaptive to changing environments without a broad base of participation providing that system with information of different types. Even more has been gained when co-management has expanded to “cooperative management” and other stakeholders, not least conservation NGOs, have become involved in management, because the competing objectives of these groups provide for a more balanced participation. However, co-management also implies a need for establishing polities as various scale levels, hence adding more layers to the management process. The objectives of the various stakeholder groups do not necessarily agree with those of the policy makers or with each other [2]. Indeed, different policy makers often have differing objectives which may change over time. Hence, co-management makes conflicts more explicit when developing management plans. In a number of cases these more explicit conflicts have resulted in better conflict management and more effective fisheries management [3].

The potential contribution of cooperative management, coupled with the conflict that is ubiquitous in fisheries management, defines the challenge for the development and ongoing maintenance of a knowledge base for effective management. In a cooperative management context, fisheries scientists are no longer granted a monopoly on identifying the truth about nature. Fishers have their own experienced based knowledge (EBK) of the resource that can complement or contradict the research-based knowledge (RBK) of the scientists. The problem of the knowledge base shifts from trying to develop objective knowledge that can be used as the basis of management measures, to developing transparent knowledge, in which each stakeholder group is accountable for explaining how they know what they know. Scientists remain in a privileged position because scientific methods are fundamentally rooted in the transparency of knowledge, but just blessing advice as objective is no longer adequate. The advice must be communicated within an environment where other knowledge claims may complement, or compete, with the RBK.

The term “scientific” is used in this chapter in its broadest context, and includes biological, economic and social analyses undertaken in support of fisheries policy. The role of science is to contribute to the development of a knowledge base from which advice on appropriate management measures can be articulated. This process happens within a highly uncertain natural, social and institutional environment. In autocratic systems, where management is imposed by professional government fisheries management agencies, fisheries scientists have only one group to communicate the results of their analyses to. In such cases, these fisheries managers often have an academic understanding of

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*lowest and enforcement costs are highest, state and common property rights will be found in the middle ranges, and individual rights will be found where unit value is high and enforcement costs are lower. Fisheries tend toward open access because the enforcement of rights tends to be costly. Boyce [58] suggests that rights-based management would favour skippers and crew at the expense of input suppliers (including boat owners), and hence pressure is exerted to maintain sub-optimal management measures.*
the fisheries science issues, and readily understand the results of the science. However, such institutions are relatively rare in fisheries internationally. More commonly, fisheries policy is developed by government agencies with a broader perspective than the fisheries themselves, and often do not have fisheries expertise (or have varying mixes of fisheries expertise). Similarly, in co-management systems, while stakeholders have considerable expertise about their fishery, they often do not have a detailed understanding of the science that underlies the advice, although cooperative research efforts have been shown to alleviate this problem considerably [4]. While the ideal goal of science is to produce knowledge that is completely transparent, the technical demands of this transparency, especially the critical reliance on quantification, makes science opaque to the other stakeholders. Furthermore, the conditions for producing scientific knowledge within fisheries management are far from ideal. Consequently, effective communication of the results of the science is essential to ensure that the advice is given the appropriate consideration. Other stakeholders, who lack the methods and tools of science face even greater challenges.

In this chapter, issues relating to developing a knowledge base and the communication of scientific advice are discussed. In particular the focus is on communication in a cooperative management situation in which all stakeholders need to understand the advice and where they may each have knowledge and perspectives of their own that can contribute to the knowledge base. We begin by describing what we see as the basic elements of the problem. The first such element is the generation of advice in the broad context of the precautionary principle. The second is the processes involved in creating a knowledge base that is common among the various fisheries stakeholder groups. Next we examine the implications of the multiple scales, multiple uses and multiple user groups found in a fisheries and the multiple costs of data gathering and analysis for the development of advice. We then turn to a series of questions about the use of models of economic and biological realities as the basis of cooperative fisheries management advice, and the use of indicators for both capturing and relaying information between stakeholders and managers. Finally, we apply the discussion to the development of management strategies that make use of harvest control rules.

13.2 DEFINING THE PROBLEM

13.2.1 The precautionary principle and the form of advice

The Code of Conduct for Responsible Fisheries [5] and the United Nations Conference on Straddling Fish Stocks and Highly Migratory Fish Stocks [6] formalized the precautionary approach to fisheries management. The basic idea of the precautionary approach is that where policy makers are uncertain about the extent of any environmental impacts from an activity then they should choose a path that reduces the activity. The precautionary approach has been thoroughly adopted in principle by the fisheries science community. In two surveys of fisheries scientists, one in the United States [7] and one in Europe [8] 80% or more agreed with the statement “It is critical that fisheries management be risk-averse and choose lower fishing pressure when stock condition is uncertain”.
This agreement in principle has lead to a great deal of effort in developing the practical implications of the precautionary approach for developing a knowledge base for management. As Degnbol [9] argues, most of this effort has gone in to finding ways to fold precaution into the pre-existing, prediction-based approach to the knowledge base, an approach he calls “stochastic predictability”, rather than examining the implications of the precautionary approach for the question of the predictability in fisheries as such. Stochastic predictability reduces the precautionary approach to something applied to sampling or other forms of measurable error, leaving aside broader questions of uncertainty ranging from bias due to model selection, through the reliability of data, the impact of ecosystem processes, and social and economic determinants of changes in fishing behaviour.

Hence a basic part of the problem of delivering complex scientific advice to multiple stakeholders is the adequate conceptualisation and communication of uncertainty. In one well-known kind of risk theory, risk is defined as the combination of three things; outcomes that affect what humans value, the possibility of occurrence, which is formally termed as uncertainty, and a formula to combine both elements [10]. Risk assessment involves three steps. The first is characterizing the depth and breadth of the potential negative effects, i.e., the implications of human interventions into marine ecosystems. The second step is to differentiate among and address separately three kinds of unknown qualities. The first is where incomplete knowledge does not allow a complete resolution of cause-effects relationships, which this approach terms as complexity. The second is limits on our ability to make inferences about future outcomes, which this approach terms as uncertainty. Finally, when the results of the assessment lend themselves to different interpretation, this is termed as ambiguity. The third step is finding ways to make decisions under complexity, uncertainty, and ambiguity [10]. The lesson for risk theory for our topic is that offering precautionary fisheries management advice requires the disaggregation of the kinds and sources of unknown qualities and an assessment of what it means to be cautious in respect to each of them.

### 13.2.2 Creating a useful common picture of the situation

The most basic requirements of various stakeholders building a common picture of the resource is that they account to each other in a transparent way for how they know what they know. There are three major issues encountered in this process. The most obvious is that stakeholders’ pictures of the resource are coloured by their own activities, interests and concerns. The second issue is putting stakeholder knowledge into a form that is transparent to other stakeholders. The third issue is putting stakeholder knowledge into a form useful for supporting management decisions.

**Varying interests and perspectives.** All stakeholders invariably select a certain set of facts as relevant. This selection of facts and the use to which they are subsequently put will create a picture of the resource that reflects the institutional imperatives that this stakeholder group is facing: fulfilling an assignment to characterise a fishery in a particular way; keeping a fishing business operating; making a decision about imposing a management measure; raising money to keep your conservation campaign alive, etc. Everyone’s tasks and problems help define their picture of nature. This issue is clearest
in respect to the fishing industry, which has the most at stake economically and which operates in different fisheries and gears and over very different geographical scales [9].

The implications of varying interests and perspectives are more controversial among scientists and managers. These groups, for example, tend to be personally committed to conservation objectives and the precautionary approach and even supportive of allowing the precautionary approach to influence scientific judgements [7, 8]. The question of to what degree scientists should advocate for specific policies is a hot topic among environmental scientists in general [11]. This debate resolves to some extent around the idea that scientists’ perceived objectivity is the basis of their role in management [12] and that they are placing this role in danger if they advocate policies. The idea that the main factor is a perception of their objectivity rather than the transparency of their methods is shown to be false in the common situation where scientists are not perceived as objective but their findings continue to play a critical legitimating role.

Putting knowledge into a transparent form. Accounting for how you know things is not just the basis of a cooperative approach to management it is also the basic principle of science. The ideal of science is replication, the ultimate transparency. The reason science relies so heavily on quantification is that the precision of mathematics provides the most transparent accounting possible. The irony is that this ideal of radical transparency, and the tools of precision it has engendered, is what has made science so opaque to the non-specialist. When living scientists attempt to realize this ideal of radical transparency they are confronted with this reality of science as a human institution made up of specialists with different specialities, skills and training. As a result the establishment of working knowledge involves many factors beyond replication, the most important of which is the culture of the relevant scientific community that determines if a new result is going to be treated as plausible or anomalous [13]. Hence, the scientific ideal of transparency is embedded in networks of trust; trust in results based on how well they fit expectations and trust in people based on judgements about their skills and qualifications.

Other stakeholders do not have a highly developed set of procedures and networks designed to answer the question ‘how do you know what you say you know’ about the condition of the resource. Conservation groups and fisheries managers usually deal with this by seeking to draw on research-based knowledge produced by others as the source of their own claims. To a growing degree this is also true of user groups as more and more fishing groups hire scientists or engage in collaborative research [4].

User groups, however, have another source of information about the resource: experienced based knowledge (EBK). EBK is produced by day-to-day activities. Fishers have extensive knowledge of the geographical range of fish populations and precise information on where and when fish congregate [14, 15]. Observers have described how EBK is used by local people in fisheries [16, 17] and many scholars believe that it should be used more [18, 19] for a number of reasons. Fishers EBK contributes to management by: (1) providing additional indices for use in stock assessments and scientific debates; (2) providing data on responses by fishers to management measures and on the status of poorly understood species (3) suggesting novel hypotheses and (4) enhancing long-term legitimacy of the management regime [20]. When user groups do not trust the data on which management decisions are based they do not cooperate and may even develop
confrontational postures [21]. From a more positive side, the user group knowledge can make the use of real time management measures feasible.

When user groups seek to bring EBK to the table in cooperative management situations they face a number of difficulties with accounting for how they know what they know. It is generally difficult for other stakeholders to evaluate EBK without a fairly broad knowledge of the local context [22]. User groups tend to view the resource on smaller temporal and spatial scales than other stakeholders [19, 23]. Perhaps the greatest difficulty user groups face in accounting for their knowledge are issues of cultural coding involving both tacit v discursive knowledge and oral vs. written knowledge; these dimensions are discussed in Chapter 4.

Placing knowledge into a form that can trigger management decisions. This tension between the ideal of science as radical transparency and necessity to act out this transparency within networks of trust creates very serious problems when scientists are called to address practical problems defined by the need for policy decisions, i.e., in arenas such as fisheries management. Scholars who have examined science in regulatory policy [24, 25] have found that the combination of high uncertainty and the demands of legal, as opposed to scientific, precision have often left scientists on the defensive. In regulatory processes it is the management institutions, not the scientists that define the question that must be answered. The reality of science as a community of trust, and the ideal of science as the source of transparent truth, are broken apart in the face of the demand for answers to specific regulatory questions that have to be justified to stakeholders. This is especially true in the most common situation where management institutions are legally constituted and subject to legal standards. ‘Adequate evidence’ is fundamentally problematic in courts because unremitting scepticism, which allows no room for questions of trust, is part of the legal process [26]. When science becomes primarily about answering legal and bureaucratic questions, its promise of transparency, the very thing that leads to it being assigned its ‘objective’ role in the first place, is placed under considerable stress.

Examples can be found in Wilson and Delaney [27] who describe influences that some of the demands for changes in the EU management system have had on the generation of formal advice and on the ways that scientist interact. One of these is a demand to shift the basic unit of reference for advice from the fish stock to the fishery. This shift has many advantages from the perspective of both managers and the industry because it links the scientific advice to the level of execution (Section 13.3.1) of many management measures. It is also, however, a requirement that biological advice be fitted to a social unit rather than a biological one. Unlike fish stocks, fisheries compete with one another and have lobbyists and politicians that speak for them. While some progress on this has been made here there are several difficulties. One is definitional. Fisheries have porous boundaries that individual fishers can cross. Another is that fisheries, as political units, demand to be treated “fairly” in respect to one another. Such fairness is a political rather than scientific value, but scientists feel pressure to make sure that they are consistent in the ways that they approach fisheries and their associated stocks whether such consistency is biologically justified or not. Fisheries-based advice makes it even more difficult to keep a clear line between science and management and places new and extensive demands on data gathering and management. Different aspects of stakeholder participation in the European Common Fisheries Policy (CFP) also lead to the somewhat paradoxical
demands that advice not be open to different interpretation but still allow flexibility in management decision-making by the various stakeholders. On the one hand, demands for accountability from the many different stakeholder groups, make it imperative that management advisors be able to point to clear results that justify their advised measures, but these results cannot be too clear, i.e., they cannot be so closely tied to the measures they are justifying that they limit managers’ options [27].

As discussed in Chapter 4, fishers face even greater challenges in respect to put knowledge in a form usable for management. Fishers focus on what is needed to catch fish, not to manage them, and the knowledge they do have is often difficult to articulate. In fisheries, users often see fisheries as systems in which small perturbations may have substantial future consequences [28] and are likely to emphasize the importance of environment over population dynamics [16, 21]. Both of these viewpoints can be difficult to fit into a picture of the resource on which management decisions can be based because such decision making requires that ecological complexity must be simplified to a point where decisions can be made [29].

13.3 THE DIMENSIONS OF THE PROBLEM OF FORMULATING ADVICE

13.3.1 Multiple scales

The scale level both of what is being managed and the institutions doing the management is a critical variable for generating an effective knowledge base. This is not “large scale” in the commonly used sense of larger fishing boats. The relevant kinds of scale are partly geographical scale, i.e., the size of the area, but the social scale, i.e., the number of people involved, is much more important. For example, pelagic fisheries are spread over a wide area of the ocean and fished by large vessels, but one important reason that they are often managed and exploited more effectively than many other fisheries is that they are small-scale in the social sense, meaning that their management and exploitation involves a relatively small group of people.

Cooperation between scientists and fishers on small-scale levels has in many instances resulted in a knowledge base for management perceived by most stakeholders as useful and legitimate [4]. Interviews with scientists in these situations indicate that fishers have begun, sometimes fairly quickly, to learn to think scientifically, in the sense of empirically and causally, about the resource and its management. Moving toward larger scales the process of building a knowledge base quickly looses these benefits from interactions. The information being communicated between different groups becomes much thinner and more highly codified. It has been extracted from local “lifeworld” [30] of background assumptions that facilitate rich communications. This idea of “lifeworld” points to the common sense idea of how the meaning of what someone says is not merely found in the words they use but in the social and cultural context in which they say them, especially the give and take of the discussion in which they are uttered. The more a communication is taken out of its context the less meaning it is able to convey. This is as much true for scientists as it is for fishers. In fisheries science, facts, models and results are also only really understood by other stakeholders when the scientists are able
to explain them. Collaborative research programmes are certainly possible on large scales, but those dealing with large-scale fisheries tend to assign fishers to specific tasks, often as data gathers or providers of research platforms, that constrain interactions [31].

Figure 13.1 outlines the problem of the knowledge base for fisheries management as it relates to geographical scale. We can assume for the sake of this discussion that geographical scale is a reasonable proxy for social scale as the two are at least correlated. Natural processes should determine the geographic extent of the management question, although this is often not true in practice because of constraints such as multiple jurisdictions. Management issues and informational needs are displayed across the bottom in relation to the scale over which they must be managed. Three institutional levels are critical. “D” indicates the level of management decision making, and this is at the highest level because decisions must be made in response to overall conditions. Knowledge for use at level D will be gathered at multiple resolutions reflecting the relevant natural processes. “E” is the level of execution of management measures; this may be a nation, a fleet, a fishery or a fishing port depending on the measure. Knowledge related to the implementation, monitoring and evaluation of measures requires resolutions at level(s) E. As management systems usually entail several management measures they will also usually entail several levels of execution as well. Finally, “I” is the level of implementation, and it reminds us that all management is finally implemented by the behaviour of individual fishers. Level I forms the lowest level of resolution, that of individual behaviours, for use on level D.

Information for ecosystem assessment, user group monitoring and control, needs to flow between these three scale levels. This involves a packaging process that is laid out in Fig. 13.2, which we call the “processes of resolution” because it is how knowledge is gathered that must be related to other knowledge generated at the same scale level to build comprehensive pictures of the situation at all relevant levels [32]. Communications about what information is needed have to move outward from the institutions responsible for the decision-making level to the users and fishers.
for the overviews at the higher scales. Communications about the situation at the lower scale levels then have to be packaged and sent inward. This packaging process involves four kinds of changes in the information (adapted from [33]): simplification to restrict the communication to what will be useful and understandable; abstraction to extract what is generalisable from the local situation; codification to place the knowledge into a symbol system that allows comparisons; and, standardization to ensure that the simplification, abstraction, and codification are done according to a common system [32].

It is within this packaging process that systematic distortions of communications [34] have their impact. We mean something very specific by the phrase “systematic distortions of communications” and the term “systematic” points to the institutional system, not merely to general regularity. This is not about mistakes or even being intentionally misleading. It is also not the packaging process itself that is the source of distortion. Systematic distortions of communications relate instead to possibilities for institutional learning. These are systematic blocks to people raising knowledge claims. What makes an institution able to learn something new begins with someone being able to point that thing out in a way that gets enough attention to bring about a change in practice. The diffusion and concentration of packaged information across scale levels is necessary for marine management institutions to work, but the information packaging will reflect institutional imperatives. These imperatives, arise from many other things besides sensitivity to new information.

If you are, for example, a scientist giving advice about the situation with some fish stock for use by managers who are trying to manage multiple stocks, you are going to have to begin by putting that advice into a particular form. That form, in many cases in Europe, is driven more by the institutional imperative to allocate fish stocks among Member States than by the needs of the adaptive management for resource conservation. You can send all kinds of text around that form that explains the problem with the information you put in it, but the management system is geared to pick up and use the packaged information, not the caveats, even if the managers would like to do so. What you really want to communicate about what is going on is systematically blocked. The
larger the scale of the institution, and the longer the jump between levels, the more difficult it is to overcome the systematic distortions driven by institutional imperatives.

13.3.2 Multiple uses and multiple users

Most analytical tools for providing fisheries biological advice implicitly assume a single, homogeneous fleet that exploits a single species.\textsuperscript{2} Similarly, fisheries management in Europe is also oriented towards single species rather than multispecies management. Further, most management is focused on a single use and user of the resource, namely the production of fish by the commercial fishing sector. However, the marine environment provides many other services that are utilised by other user groups. These alternative uses of the resources require additional information to be considered in the development of fisheries management plans.

Even considering only the commercial fishing industry, there is considerable heterogeneity in terms of resource use and the set of user groups. Most fisheries involve several different gear types capturing differing combinations of a set of common species as well as a set of differing species. These groups interact to different degrees, often creating conflicts. For example, mobile fishing gears such as beam trawl damage static gears such as pots or the target species of one gear type is a bycatch of another gear type, and so on. The English Channel provides a good example of the type of complexity inherent in many European fisheries. Around 4000 registered boats operate in the fishery ranging in size from 4 m to over 30 m. These boats catch over 100 different species, with 40 species, from 53 different stocks making up the bulk of the catch by value [35]. In total, 72 separate fishing activities have been identified in the Channel [36], catching different combinations of the main species. The fishing activities are broadly based on seven main gear types: beam trawl, otter trawl, pelagic/mid-water trawl, dredge, line, nets and pots. While fleets can also be broadly classified on the basis of their main gear type, they are largely multi-purpose, and operate in several different métiers over the year. For example, many beam trawlers also operate using otter trawl and dredge during part of the year, while the inshore mixed sub-fleet undertakes diverse range of fishing activities, switching from gear to gear on an opportunistic basis.

Biological and economic advice often differ in the way that the multiple users and uses are considered. While biological advice tends to focus on the individual species within the fishery—species that may be exploited by several fleets—economic advice often focuses on the individual fleets (which themselves may target several species). For example, SGECA [37] estimated the economic impact of the set of total allowable catches (TACs) recommended by the scientific advice on the different European fleet segments. Such advice tends to be reactive rather than strategic. That is, the economic advice does not, in practice, propose an alternative set of TACs that may achieve bioeconomic objectives, although the provision of such advice is feasible.

\textsuperscript{2} An exception to this is the 4M model that considers multiple species, fleets and gears [59], based on multispecies VPA. These models, as well as the multispecies VPA analyses, are still largely considered experimental and are not formally used for policy advice.
Users of fisheries resources extend beyond the commercial fishing sector. Other stakeholder groups with an interest in fisheries include the recreational fishing sector, the processing sector and environmental groups. Marine tourism also has an interest in the effects of fishing on the marine environment, e.g., damage to reefs may reduce benefits to scuba divers. Extractive industries such as oil and aggregates also interact with fisheries through their use of common areas. As will be illustrated in the following section, these groups have different objectives and will aim to influence fisheries management to achieve these objectives.

The incorporation of these additional interest groups into the scientific and economic advice provided for management varies considerably. For some species, such as sea bass, recreational catch is a major component of fishing mortality so estimates of recreational fishing activity are included in the scientific advice. In most instances, however, these groups are not routinely considered in the production of advice, and their input into the management process has largely been through lobbying and the production of “independent” studies that are commissioned by the individual groups. Assimilating this additional information with the fisheries oriented scientific and economic advice can further complicate the management decision-making process.

13.3.3 Multiple objectives

As noted in the introduction, fisheries management is often characterised by multiple objectives. In many countries, these include conservation, economic and social objectives, while in some countries foreign exchange earnings and food security are also important when determining management strategies. These different objectives reflect the multifunctionality of fisheries such as wealth generation, employment and sustenance.

Within Europe, fisheries management objectives have been found to vary not only from country to country, but also from fishery to fishery [2]. Numerous methods exist to incorporate the existence of multiple objectives into fisheries policy advice (for a review of such methods, see [38]). However, a key difficulty for the estimation of management advice is that these objectives not only differ from fishery to fishery, but the importance of the objectives varies between the different stakeholder groups within the fishery. As a result, identifying a single “optimal” fleet composition or TAC is not feasible, as the concept of optimal will vary from one stakeholder to the next.

An example of the difference in importance of the different objectives for a UK fishery is demonstrated in Fig. 13.3. Four separate stakeholder groups in the fishery were identified. The importance of some objectives varied considerably between the stakeholders groups, particularly those relating to environmental impacts and bycatch. Similarly, the importance of maintaining employment and profitability also varied.

Such diversity in priorities helps to explain the existence of conflict between the various stakeholder groups. Fishers are more likely to oppose management advice that results in improvements in environmental conditions at the expense of employment or profitability, while environmental groups will oppose advice that leads to the reverse, i.e., higher employment and/or profitability at the expense of reduced environmental conditions.
13.3.4 Multiple costs

The collection and analysis of information in order to provide policy advice is not costless. While increased information reduces uncertainty and therefore results in more robust management advice, the marginal cost of collecting additional information will at some point outweigh the benefits in terms of reduced uncertainty. Consequently, we can consider there to be an “optimal” amount of uncertainty at the point at which the marginal cost of reducing uncertainty balances the marginal benefit of the reduced uncertainty (Fig. 13.4).³

The exact level of optimal uncertainty will vary from fishery to fishery depending on the ease at which information can be collected and the complexity of the environmental, biological, economic and social aspects of the fishery system. In practice, identifying such a level is not feasible. Indeed, attaining certainty may also never be feasible. Improving information often leads to the detection of new uncertainties, so perfect information, and thereby certainty, may never be attained [39].

When fishers are not required to pay for the costs of the information collection, the marginal cost to the industry of obtaining additional information and reducing uncertainty is zero. Consequently, the optimal level of certainty for the industry is complete certainty or as near to complete as is possible given the state of the information gathering and

³ A review of the issues relating to the estimation of marginal benefits and costs of information for resource management is provided by Caughlan and Oakley [60].
As a result, industry will continue to insist on more information being collected and lower levels of uncertainty before accepting the validity of management advice. In contrast, when the industry is required to pay or contribute to the costs of management, as is the case in, e.g., Australia, New Zealand, Iceland and Canada, advice with lower levels of certainty is acceptable provided industry have a say in the level of research that they are paying for.

13.4 MODELLING REALITY AND THE FORMS OF ADVICE

13.4.1 Dealing with complexity and uncertainty in the provision of advice through models

The previous section highlighted the considerable complexity in fisheries that make the provision of advice difficult both in its formulation and communication. In addition to this complexity, considerable uncertainty also exists. Rosenberg and Restrepo [40] identified five main types of uncertainty that affect the accuracy of fisheries management advice. Measurement error arises through errors in the data used in the analyses, ranging from deliberate misreporting of information by fishers to non-deliberate sampling errors.

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4 A similar argument holds for the demand for fisheries management services in general. However, in this case, the demand is not likely to come from industry (who may in fact see management as an intrusion rather than a benefit) but rather other stakeholder groups. These may include government ministries and agencies responsible for management themselves who directly benefit from increases in management services [61].
Process error occurs as a result of natural variability in the biological and economic system (e.g., environmental fluctuations that result in divergences between “predicted” and “actual” outcomes). Estimation error involves biological or economic parameters being incorrectly estimated and subsequently used in the analysis to develop the advice. These can arise through measurement error, but can also arise as a result of errors in the estimation process, such as through incorrect assumptions about the relationships between the variables within the system being estimated. Different assumptions may result in different parameter estimates and subsequently different advice. Modelling error involves incorrect specification of the model. For example, the true dynamics of the system may differ to those included in the model as a result of imperfect knowledge of the true system. Finally, implementation error occurs when the industry responds differently to a given scenario than is assumed in the model.

The major difficulty facing all fisheries scientists arising from these different types of uncertainty is that there is no way of knowing the “truth”, and no way of knowing how close to or far from the truth we are. That is, we do not know what we do not know. When developing models for fisheries analyses, assumptions are made about how the system behaves (the assumed world), and a model is developed that tries to incorporate these relationships. In many cases, not all relationships that are believed to exist can be parameterised, and tradeoffs between model complexity and robustness need to be made (see below). As a result, the model is a simplification of the assumed world. However, we do not know if our assumed world is a good representation of the real world, which may contain relationships that we may not even be aware of, such as in Figure 13.5(a), or if the real world is considerably different to our assumed world (Fig. 13.5(b)). Box [62] summed up this problem as “all models are wrong, but some are useful.”

The relationship between the model and the assumed world is a function of the level of complexity that can be captured and parameterised in the model. However, increased complexity does not always result in improved model performance and usefulness for management advice. Models that are highly complex require many parameters, estimates of which will contain their own uncertainty. Costanza and Sklar [41] and Håkanson [42] suggest that the relationship between complexity and performance is parabolic (Fig. 13.6). Too much complexity leads to too much uncertainty and problems with interpretation of

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**Fig. 13.5.** (a) Assumed world is representative of the real world. (b) Assumed world is unrepresentative of the real world.
the model’s dynamics and predictions, while too little detail results in models that cannot produce realistic behaviours. Thus, there may be an ‘optimum’ level of model complexity and this may be substantially below the maximum possible. Indeed, complexity introduced for the sake of completeness may be counterproductive if the resulting model is actually of poor quality. The key challenge facing modellers is therefore in striking a balance between complexity and uncertainty.

The trade-off between complexity and uncertainty is also complicated by stakeholder belief in the models.\(^5\) Stakeholders have highly detailed knowledge about different components of the fisheries that they consider essential to include in the analyses. Excluding such details, while potentially improving the overall robustness and usefulness of the model, reduces the stakeholder acceptance of the results. In contrast, including the detail, results in greater uncertainty in the model results, thereby reducing the usefulness of the model for decision-making. As the true state of the resources can never be known (unless they are driven to extinction), there is no objective measure against which to assess the consequences of using simple rather than detailed models.

The above relationship is equally applicable to both biological and economic advice. An observation noted by a PhD student working with both economists and biologists, however, is that biologists tend to prefer greater complexity, whereas economists tend to prefer less complexity.\(^6\)

\(^5\) This is based on the outcome from a series of workshops on biological and economic uncertainty in fisheries policy advice [64].

\(^6\) J. Ramon, IPIMAR (Portugal), personal communication, March 2005.
13.4.2 Communication knowledge and advice using models

A major factor contributing to fisher compliance with fisheries regulation is a feeling of legitimacy of the policy (Chapter 14). While this legitimacy includes perceptions of social justice, it also includes belief that the management is based on reliable advice. From the above, it is clear that fisheries advice never will be, nor can be, certain. However, robust policy advice can still be developed with uncertain information. Communicating the robustness of the advice to industry and policy makers is therefore essential to gain stakeholder confidence. However, communication involves more than just dissemination or transfer of information, it involves transfer of understanding. This requires a two-way process rather than a one-way flow of information [43].

One method that has proved successful in a number of natural resource based industries, including fisheries, is to involve fishers in the development of models and the evaluation of policy advice using these models. This has been found to enhance mutual understanding and consensus between stakeholders, as well as encourage stakeholders’ accountability and commitment to the final decision [44]. Stave [45] found that stakeholders do not need to know anything about modelling to be able to gain understanding from the participation in model development and use.

Stakeholders can provide direct input into bioeconomic models through highlighting areas of major uncertainty and providing information on expected outcomes based on their own experiences. Examples of models constructed solely on the base of stakeholder-provided information already exist. For example, Scholz et al. [46] developed a GIS model for assessing marine protected areas based solely on stakeholder knowledge. The information derived from the fishers’ knowledge corresponded closely to those derived from more traditional scientific approaches (e.g., fisher’s perceptions of the stock status were similar to those derived through formal stock assessments). In other models, stakeholders have been able to provide information about specific components of the system that would otherwise have been generalised [47].

The model development must also take into consideration the ability of the stakeholders to use the model. Connors [48] found that industry’s reluctance to engage in the modelling process was partially a consequence of the complexity of the model. Development of user-friendly interfaces has been undertaken for a number of other natural resource models with the particular aim of facilitating stakeholder involvement [49]. Ideally, the model design should be sufficiently flexible to enable stakeholders to run a wide range of alternative scenarios without the need to understand the modelling language. The use of graphic user interfaces—both for input to and output from the model—is one way in which stakeholders can readily run different simulations and see the results. Data visualisation has proven effective in communicating the results of complex models in other industries [48]. In such cases, trade-offs between the multiple outcomes of different policy scenarios can be readily seen by stakeholders, enabling informed decisions to be made. Increased stakeholder use of models will require increased emphasis on model appearance and usability as well as robustness and scientific rigour in model construction.

A key advantage of using models as a means of communicating with stakeholders is that all assumptions are explicitly presented. Without a formal model structure, misperceptions and different assumptions regarding the key relationships underlying the policy
advice may result in industry, government and scientists talking at cross-purposes. With a formal model, the assumptions themselves are explicitly represented, and discussions can focus on the appropriateness (or otherwise) of the assumptions. Consensus on the model structure and assumptions will result in more readily acceptance of the model results by all parties. Using a model as an interactive forum can therefore foster common understanding about the system and consensus about management actions [45]. Participatory approaches to modelling have also been found to reduce criticism about the use of models in natural resource policy advice where the assumptions and uncertainties underlying the assessments had otherwise been difficult to communicate [47].

13.4.3 Use of indicators for communicating and relaying information

The focus above on the production and use of quantitative models does not diminish the role of the use of indicators and qualitative analysis both in the production and provision of scientific advice. In many instances, quantitative economic analyses are supplemented by more qualitative analyses that use theoretical (rather than numerical) models of behaviour to determine the general direction of an impact rather than an absolute magnitude. These qualitative analyses consider the incentives generated by different management systems, economic conditions and belief in the state of the resource to provide guidance on the types and broad magnitude of outcomes that may be expected rather than specific levels of outcomes. The qualitative analyses, by their very nature, are imprecise, but the general direction of outcome (e.g., increase, decrease or stay the same) may be more robust than complex quantitative analyses that are sensitive to small parameter changes. Further, the qualitative analyses are able to incorporate qualitative information that is not readily parameterised, and hence excluded from the quantitative models.

Indicators can be used to both capture knowledge from industry as well relay information. For example, simple economic indicators can provide information on fishers’ confidence in the state of the resource and effectiveness of management. In particular, the market trading prices of licenses, quota and vessels all provide information on fishers’ expectations on the future profitability of the fishery. The value of licenses and quota reflect the discounted expected flow of profit over time. Where the license is not separable from the vessel, then this value is also reflected in the vessel price [50]. Increases in price reflect expectations of higher future benefits even if current profitability is low. Conversely, declining prices reflect expectations of declines in profitability, either through a decline in the resource base or ineffective management policies. These are imprecise estimates as they are affected by management constraints (e.g., difficulty or ease of transfer, degree of divisibility) and the level of compliance [51]. However, they are useful qualitative indicators of expectations and relay the knowledge of the industry to managers.

Qualitative indicators as well as the results from quantitative analyses can be combined into decision rule systems. These are effectively non-predictive adaptive approaches, as they do not aim to predict the results of certain management measures. Instead, these systems focus on monitoring the system (in a broad sense) and adapting to developments and changes, which are discovered by means of generally agreed indicators on the state of different elements of the system. In some cases, these indicators may be the result of
quantitative analyses. This approach is being implemented in Europe for the first time in the current recovery plan for Southern Hake.

The best known example of the non-predictive adaptive systems is the so-called ‘Traffic Light method’, which has been applied in the advisory process for the Northwest Atlantic shrimp stock and on trial basis for some groundfish stocks in the Scotia-Fundy region, Canada. The basic element of this method is a broad range of indicators, which represent estimates of certain attributes of the fish stocks and the fishery. These indicators, which must be carefully described, validated and generally accepted by the concerned interests, can be categorised as stock assessment indicators, indicators of ecosystem effects of fishing, indicators of economic and social outcomes and, finally, indicators of regulatory compliance. Reference points are determined for the state of each of these indicators to determine the boundaries between the traffic light colours: green (go ahead), yellow (beware) and red (danger). The traffic light element of the method is mostly for communication purposes and any number of colours, scaling or just numbers could be equally suitable; the original indicator values should always be accessible to the decision-makers. Different, changing combinations of red, yellow and green in various categories should then ideally lead to specific reactions, which have been negotiated beforehand so that the measures can be applied immediately and the system in this way adapt to the changing conditions; this is the element of decision rule system in this method. The form of decision rules is yet to be developed for this system but it is predictable that they will relate to similar measures as harvest rule systems.

13.5 THE DEVELOPMENT OF FISHERIES MANAGEMENT STRATEGIES

Harvest control rules (HCR) aim at reducing the reliance on political processes in decision-making on management measures. Harvest control rules are attempts to create self-binding mechanisms that seek to protect the fisheries management system from ongoing pressures to meet short-term needs at the expense of long-term benefits.

Interest in harvest control rules has been growing in European fisheries management. One reason may be that in the midst of cross pressures from fishers and conservation groups, managers see these as a clearer source of justification for their decisions. Historically, TACs have been negotiated on a yearly basis and the introduction of harvest control rules has been linked to multi-annual fisheries recovery plans and to a lesser extent multi-annual management plans (Fig. 5 and 6 [52]). These recovery plans were an important part of the provisions of the new basic regulation of the CFP that was implemented from 1 January 2003. Under these harvest control rules one or more target points are set for stocks that are considered to be outside safe biological limits. These target point are reached over a period of time, by setting, as only one example, a TAC in accordance with a pre-defined fishing mortality rate. HCRs in Europe mainly consist of two preset sizes of spawning stock biomass (SSB) with fixed levels of fishing mortality to take effect when the stock is above the larger or below the smaller size, with a smooth transition rule between the two sizes [53]. The first species to become subject to a recovery plan in the EU was cod in the North Sea [54].
One way of thinking about HCRs is as perhaps the ultimate expression of the “management as a technical problem” approach; the political is simply ruled out. This is a mechanical conceptualisation of management where a scientific evaluation is plugged in and a management decision pops out. Science is completely separated from management, but the real decisions about outcomes are made on the scientific side. It is an approach that can be very attractive to scientists and managers [55] as well as to fishers, to the extent that it is attached to long-term goals and facilitates business planning.

An adequate understanding of harvest control rules from the perspective of the knowledge base is greatly facilitated by placing HCRs in the context of management strategies, the approach taken by ICES through the Study Group on Management Strategies [53]. They place harvest control rules within a hierarchy of decision making, arguing that such rules are always present if only implicitly [53]. Their schema is pictured in Fig. 13.7. The suggestion that HCRs are always present within a management strategy suggests that management strategies should always be concretely expressed by a series of if . . . then, state . . . response rules. The ‘strategic’ element becomes understanding how the rules fit
in the broader social, economic and ecological context so that they can be designed and implemented effectively. Figure 13.7 points at the need to rationally organize a series of decisions, and the information needed to make those decisions, within in a broad context in order to arrive at the most effective HCRs. The management strategy approach still treats fisheries management as basically a technical problem, but it makes it a more comprehensive, broadly defined and nuanced technical problem.

The problem with thinking of fisheries management as a technical problem, of course, is that politics cannot be simply ruled out. All stakeholders may benefit from a smoothly operating HCR, but they are not going to agree so easily on what that rule should be. So it is important when considering Figure 13.7 to also consider Figure 13.8. Figure 13.8 places Ostrom’s [56] hierarchy of rules for commons management into the fisheries systems context. Fisheries are one type of commons, which is a general term for goods owned or otherwise controlled by some group larger than an individual. In her system, constitutional choice rules are rules about who can make decisions, collective choice rules are rules about how and when decisions are made, and operational rules are the day-to-day rules for commons management—in our context they are the harvest control rules. The two figures are clearly linked. Making a decision about who participates in decisions is closely associated with making a decision about what knowledge is relevant. So are objectives and performance criteria, which indeed may contain participation-related objectives and criteria.

The two hierarchies, however, are quite different.

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**Fig. 13.8. Ostrom’s hierarchy of rule types.**
Figure 13.7 defines a set of rational decisions and the basic order in which these decisions have to be taken, i.e., from the outside inward to a final expression in a harvest control rule. It also describes the environmental parameters (both social and ecological) that must be considered at each step to make the decisions effective. The figure is based on a substantive, goal-oriented rationality in order to implement management measures. Knowledge is the basis on which these goal-oriented decisions are made and the demand made on knowledge is that it be objective.

Figure 13.8 also describes a set of decisions and the order, from outside in, in which they should be made. But here the decisions are not based on a substantive, goal-oriented rationality but rather on a process rationality oriented toward maintaining the ability of the stakeholders to continue to coordinate their actions to achieve ongoing, adaptive management. Knowledge is the shared picture of reality on which decisions are made and the demand made on knowledge is that it be transparent.

Effective management strategies must be developed in respect to both of these kinds of rationality if they are to be continuously effective.

Examining the scale dimension of the advice problem, the main knowledge issue on the side of implementing management measures, is collecting information in a useful form across broad areas and collating it into a form that allows decisions to be made. The main scale issue on the side of achieving ongoing, adaptive management is developing nested decision making systems [56] that ensure that decisions are made on the smallest effective level, because the smaller the level the richer and more detailed will be the useful and non-distorted information that can be applied.

Examining the multiple use dimension problem, the main knowledge issue on the side of implementing management measures is the appropriate unit of analysis. Should the analysis be considering commercial fishing fleets, all fishing effort, stocks, seascapes, tourist businesses, etc? How can all these different uses be monitored, weighted and compared with one another to assess the impacts that the measures are meant to manage? The main issue from the side of ongoing adaptive management is finding the best ways that the interests of the many different uses of the resource can be represented in the creation of the knowledge base. The people involved in the various uses can be involved in adaptive management in many ways, from participation in identifying the core findings on which management should be based to creating independent studies and critiques. Which users have the information management needs to respond in a timely manner to new threats?

Examining the multiple objectives dimension, the main problem on the side of implementing management measures is conflicting objectives: management impacts that achieve one objective at the expense of another. This leads to a strong temptation for the management system to measure only those objectives that it is prioritizing, a good reason why economic data lags behind biological data in fisheries management. From the perspective of ongoing adaptive management, the problem is the role that objective setting should play when in principle, the short and medium-term management goals will shift with changing circumstances. Is it possible to state objectives, and their corresponding indicators, that are abstract and general enough so that they continue to be valid as circumstances change, while still being concrete enough to be measurable?
Finally, examining the multiple costs dimension, the problem from the side of implementing management measures is the overall costs benefit of the information needed for the implementation of the measures. What is the level of uncertainty under which management can function without costing more than it is worth? From the perspective on ongoing adaptive management the issue is who will pay for the knowledge base for management and how are the costs to be sustainable.

13.6 CONCLUSION

Complexity in fisheries advice is a function of complexity in the natural system being exploited, and complexity in the structure of the sectors using the resource. The unseen nature of the resource results in uncertainties for their assessment that are not present for terrestrial resources. Differing objectives of the multiple users of the marine resource further complicate the management process, with uncertainty used as an argument by some groups to decrease catches, while other groups use the same uncertainty as a reason to increase catches.

Reduced uncertainty can be achieved through greater investment in research, although at some point the marginal cost of collecting the additional information exceeds the benefits of its use in the management decision-making process. As the stakeholders are not responsible for these costs, however, stakeholder groups are often not content with accepting uncertainty and demand greater precision in the advice that can be produced by biologists, economists and social scientists. Similarly, stakeholders are often not convinced by simple models used for the production of advice that ignore much of the complexity that they perceive to exist in the fishery. However, introducing greater complexity into these models may lead to the results being less robust due to the cumulative effects of uncertainty.

Models used for the generation of management advice can be used as communication tools with stakeholders. An advantage of a formal model structure is that the assumptions are transparent, and the effects of uncertainty on the results can be tested. Stakeholder knowledge can also be incorporated into the models through developing scenarios, assisting in the development of model structures, provision of data and validating and developing the key assumptions.

The communication of advice does not necessarily have to rely on point estimates of key model results. Instead, indicators can be used to relay advice, such as the traffic light approach described above. An advantage of such an approach is that it implicitly recognises uncertainty in the model results, but still allows the overall implications of the advice to be relayed to stakeholders. Indicators can also be used to capture industry expectations about the state of the resource and future conditions in the fishery.

Pulling together a knowledge base for ongoing adaptive management requires attention to the way the system is designed: what information is needed for the ongoing implementation and evaluation of the management strategy, how is this information to be collated, coded and communicated, from and to whom, when and at what cost. Where multiple stakeholders are involved, as they always are, ongoing adaptation to changes in the ecosystem and social system requires more than design. It requires a process
of participation and decision making in which participants are able to hold each other accountable. The role of the knowledge base is to provide the information that makes such accountability possible. This begins with stakeholders being accountable for providing the information needed for ongoing management.

REFERENCES

Wilson and Pascoe


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14.1 INTRODUCTION

Most fisheries are subject to regulation in one form or another. In general, fishery regulations are intended to achieve particular social objectives which would not result were individual fishing firms allowed, freely, to pursue their private objectives. Such social objectives include controlling the level of catches in order to ensure that the fishery is sustainable into the future and (in some cases at least) the achievement of economic efficiency in fishery exploitation.¹ Note that for practical or ideological reasons many fisheries are regulated with the sole objective of preventing (further) resource depletion and in such cases the regulatory instruments in place may actually lead demonstrably to economically inefficient outcomes.² In many fisheries there are also regulations that are put in place in order to achieve non-economic or distributional objectives, the resolution of conflicts between different user groups, for example [see, e.g., 1, 2]

Clearly, achievement of the intended objectives of regulations depends crucially on fishers’ compliance with those regulations. Misperceptions by fisheries managers of compliance behaviour of fishers may lead to the actual outcomes of policies differing substantially from the intended outcomes. Further, non-compliance distorts the information flow between the industry and scientists. In this chapter, we consider the issue of regulatory compliance in fisheries. We also consider how such compliance might be incorporated into a bioeconomic models in order to assess the likely outcomes of changes in management regimes.

¹ The usual economic justification for regulation in a fishery is to correct the inefficiencies which arise from the lack of property rights in the fish resource and hence an appropriate access or use price. As a result of uncorrected external costs associated with fishing, there is, generally, an excessive level of exploitation and a consequent net economic loss to society. Specifically, in the current period external costs may be imposed on other (all) fishing firms in the form of reduced harvest rates (increased unit costs), while in a dynamic setting external costs may be imposed on society in future periods due to a reduced stock. See, e.g., Clark and Munro [99], Clark [100, 101], and Arnason [102].

² For a discussion of the types of ‘traditional’ fishery regulations and their economic implications see Crutchfield [103], Scott [104], and Townsend [105]; also Anderson [106].
Regulation can be imposed at different levels. The political/institutional framework responsible for designing and implementing regulations may involve the fishing sector itself, so that there is a degree of self-regulation, or ‘co-management’ in cases where responsibility is explicitly shared between a public body and a sectoral organisation [3]. Involvement of the fishing industry in the process of regulation has the potential to reduce costs in a number of ways. Costs of information acquisition may be substantially reduced since the industry has access to private information that is costly for an external regulator to obtain. More generally, administration and enforcement costs may not only be reduced but internalised. There may be benefits from improved compliance with regulations, since regulations designed and implemented with industry involvement may be viewed as having greater legitimacy (see below) than regulations which are imposed with no industry involvement [4–10]. On the other hand, a possible danger of some degree of self-regulation is that regulation may become ineffective due to a lack of enforcement effort. This would, in effect, be an extreme example of ‘regulatory capture’ where regulation is subverted in the interests of the regulated.

In many countries the administration and implementation of fishery regulations, including monitoring and detection activities, is the responsibility of government and/or a governmental or quasi-governmental agency, whereas the imposition of sanctions for non-compliance is a matter for the courts under the provisions of criminal law. Aside from any practical implications for the costs of enforcement, one consequence of this is that, used to dealing with crimes in the wider context, judges and magistrates may find it difficult to reconcile infractions of fishery offences, for example, with the idea of crime as a wrongful act in the moral sense. This may result in a reluctance to sanction effectively. The likelihood of regulatory failure is exacerbated by the requirement for normal judicial standards of proof of both act and intent before a sanction is imposed [11]. More generally, the reliance of regulation on the legal process means that it becomes inextricably linked to the principles that underlie the law. There is often a strong ‘normative’ foundation to the law, emphasising procedural fairness and political legitimacy, for example, in contrast to a purely functional/instrumental basis which would focus more clearly on the achievement of public policy goals [12].

14.2 NON-COMPLIANCE, THE KNOWLEDGE BASE AND FISHERIES POLICY FORMULATION

Non-compliance with fisheries regulations has two potential impacts on policy formulation and its related knowledge base. First, the actual outcomes from a management policy are likely to differ from the intended outcomes assuming perfect compliance. Second, the information on which policy is based is distorted. In particular, it has an impact on the estimation of the biological state of the resource. Both aspects may potentially result in the formulation of inappropriate policy advice. In the former case, an alternative management system may have resulted in outcomes more in line with the objectives of the policy makers. In the latter case, controls imposed through management may be either too lenient or too strict. For example, catch limits may be inappropriately set as the perception of the state of the resource differs to the true state.
At this point it is worth considering non-compliance vs. compliance having negative consequences. An example of the latter is catch discarding, which, under most management regimes, is either tolerated or even required by rules on quantitative landings limits such as quotas. Similarly, catches of fish smaller than a minimum landing size are also required to be discarded. However, from the perspective of the knowledge base, discarding creates the same problems associated with illegal landings. As relatively few discarded fish tend to survive, discarded overquota or undersized fish represent fishing mortality in excess of what is observed by the regulator (i.e., landings) in the same way as illegally landed fish. From a fisheries management perspective, there is little difference if overquota or undersized fish are landed illegally (non-compliance) or discarded (compliance). The economic incentives underlying discarding are well documented in the literature [13, 14]. Fishers will continue to operate even if the quota of some species has been filled so long as the value of the catch that can be legally landed exceeds the additional costs of continuing to fish. Similarly, fishers have an incentive to use smaller mesh sizes and to discard undersized fish if the use of larger mesh sizes also reduces the catch of fish above the legal minimum landing size.

The impact of non-compliance on the knowledge base varies with different management regimes. Under output controls, the lower apparent landings per unit of effort (as not all landings are declared) result in underestimates of the size of the resource. This in turn may lead to lower quotas in the following year. Perceptions by the industry that management is based on incorrect information reduce their willingness to comply, exacerbating the problem. While similar problems exist as a consequence of discarding, attempts can be made to estimate the level of discarding for consideration in stock assessments (e.g., through observers on a sample of vessels). Similar estimates cannot be made for illegal landings, as it is likely that the fishers will comply if they know they were being observed.

With input controls, non-compliance is not likely to result in illegal landings. However, as will be further discussed in Section 14.5, incentives are created to substitute uncontrolled inputs for restricted inputs. Nominal measures of fishing effort (e.g., days fished) become distorted. Effective fishing effort (i.e., the “real” impact of the fishing fleet on the resource) may increase, resulting in increased output, even though nominal effort levels (i.e., the observed measure of effort) are maintained or even fall. As a result, catch per unit of nominal effort may be seen to increase, which may be interpreted as an indicator that stocks are improving. However, both stock levels and catch per effective effort may, in fact, be declining.

The focus of this paper is on fisher compliance with regulations. However, non-compliance by management agencies of Member States in the collection and provision of fisheries data also has implications for the knowledge base. A recent review of compliance with EU data regulations found considerable variability in compliance between Member States. This compliance varied with both provision of information requested, as well as in quality of the data submitted [15]. Variability in the quantity and quality of data provided for fisheries analyses erodes the reliability of the fishery assessments, further reducing the likelihood of compliance by fishers themselves with the resulting regulations.

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3 The relationships between compliance and the management system are more fully discussed in Section 14.5.
14.3 MODELS OF INDIVIDUALS’ BEHAVIOUR AND NON-COMPLIANCE

An understanding of the reasons why fishers do not comply with policy regulations is an important part of the fisheries policy knowledge base. Foremost, it allows for the possible extent of non-compliance to be estimated given the set of environmental, economic and social conditions under which the policy is being imposed. This enables better estimates of the outcomes from management policies to be derived a priori, enabling more reliable policy evaluations. It also enables consideration of factors that could improve compliance during the policy formulation process.

According to the standard (neoclassical) economic model of rational behaviour, individuals maximise their utility subject to constraints and firms (whose decisions are controlled, we assume, by utility-maximising individuals) maximise profits, again subject to constraints. In situations of risk or uncertainty where individual outcomes or states can be assigned a distinct probability of occurrence, in the rational model expected utility is maximised.

Based on this utilitarian model of behaviour, economic models of crime have been developed which focus on the expected gains and losses associated with a rational choice between committing an offence and not committing an offence [16–26]. This type of deterrence model is generally associated with Becker [22] and is widely used to model compliance with regulations, including fishery regulations (see below).

A single event version of the rational deterrence model can be written

\[ EU(W_t) = pU(W_0 - L) + [1 - p]U(W_0 + G), \] (14.1)

where the von Neumann–Morgenstern utility index \( U(.) \) is a strictly monotonically increasing function of wealth \( W \), \( EU(W_t) \) is the expected utility of wealth \( W_t \) if an illegal act is committed, \( W_0 \) is the individual’s initial wealth and \( G \) is the expected gain from committing the illegal act. The expected probability of being caught and sanctioned if the illegal act is committed is \( p \) in which case there is an expected loss in wealth of \( L \) (an expected monetary fine, for instance). The strictly rational individual will commit the illegal act if in Eq. (14.1) we have the total condition \( EU(W_t) > U(W_0) \) and will be indifferent between acting legally or illegally if \( EU(W_t) = U(W_0) \). Deterrence therefore requires \( EU(W_t) < U(W_0) \).

Many regulatory compliance decisions are better modelled in terms of continuous choice over a decision variable that is legal over some range and illegal above a certain threshold. For example, we could have

\[ EU(I) = pU\{B(x) - L(x)\} + [1 - p]U\{B(x)\}, \] (14.2)

where income \( I \) is the sum of a benefit function \( B(x) \) associated with an input or output \( x \) and a loss function \( L(x) \) associated with the same input or output when \( x \) exceeds a given value (an output quota, perhaps, or a limit on the amount of an input which may be used). Assuming that the benefit and loss functions converge at some feasible level of \( x \),
the necessary first-order condition for a utility-maximising solution to the choice of \( x \) in Eq. (14.2) is

\[
B'(x^*) = \frac{pL'(x^*)U'(B(x^*) - L(x^*))}{pU'(B(x^*) - L(x^*)) + [1 - p]U'(B(x^*))}
\]

(14.3)

which appears complex but collapses to

\[
B'(x^*) = pL'(x^*),
\]

(14.4)

more recognisable as the equality of marginal benefit and expected marginal loss, in the case of a constant marginal utility of income \( U'(.) \), or, equivalently in this model, if we simply maximise expected income itself instead of utility of income. Condition Eq. (14.4) is the basic marginal decision rule for a rational agent maximising over a continuous choice variable with an enforced constraint, but the total condition from problem Eq. (14.1) must still be met in order for the rational agent to be deterred from acting illegally within a given period. The Principle of Marginal Deterrence requires that expected penalties are always cumulative at the margin, in order to obviate the perverse possibility that there is a net benefit in committing larger rather than smaller violations [27].

The essential policy prescriptions that stem from this type of model relate to the optimal choice of the probability of detection and sanction, which depends upon enforcement effort, and the size of the penalty. Since the expected penalty is a product of the (subjective) probability of detection and the penalty if sanctioned, and since enforcement is usually costly, the general conclusion is that deterrence should as far as possible be achieved by increasing the penalty [28–30]. Ehrlich [19] considered the possibility that \( p \) was related to the amount of time spent by the individual in illegal activity, i.e., habitual offenders have a biased assessment of their risk of getting caught. Thus habitual offenders may have a lower assessment of the risk of detection, for example because they have developed techniques for avoiding detection, or because they have been lucky enough to avoid detection in the past. Alternatively, it is plausible that persistent offenders have a higher assessment of their risk of getting caught, because they know that enforcement is being targeted at them, or perhaps because they think they are ‘running out of luck’.

This basic approach has been widely used in order to model regulatory enforcement and compliance in environmental management problems such as pollution control [31–37] as well as in the regulation of commercial fisheries [38–42].

Implicit in most economists’ rational model of behaviour is that preferences are exhibited over, and utility derived from, only those ‘goods’ which can be consumed in the ordinary sense, or, indirectly, from the income or wealth that enables consumption of such goods. Some modifications or extensions to the strictly rational maximising or optimising model of economic behaviour do not fundamentally challenge the standard utilitarian assumptions. Risk aversion, for example, is often modelled by assuming a diminishing marginal utility for wealth or income, or with a utility function which is sensitive to the variance of expected wealth or income [43]. In either case, the utility function is assumed to have some particular form without any explicit modelling of what produces that form.

Theories of ‘bounded rationality’ and ‘satisficing’ [44–46] focus principally on the idea that individuals have limited access to, and limited ability to process, information
relevant to their choices, so that the model of rational ‘optimisation’ is unrealistic in describing both process and outcome in decision-making behaviour. Related theories argue that the ability to optimise is also compromised by uncertainty [47, 48]. These and similar ‘cognitive’ theories of decision-making allow for non-cognitive influences such as emotions or behavioural norms as guides or ‘stopping rules’ (heuristics) in the decision process, but they remain essentially theories of utilitarian rational choice [49, 50].

In the non-economic social sciences more attention is devoted to the influence of personal and social norms, social influences of a more instrumental nature, as well as more or less contemporaneous judgements about the ‘legitimacy’ of regulations, in determining individuals’ behaviour under regulation. Although theories of behaviour in sociology and social psychology place considerable emphasis on such factors, there is no conventional or generally accepted economic model of behaviour that explicitly incorporates moral norms or other behavioural norms, for example.

Norms may be defined as expectations about an individual’s own actions, or those of others, in terms of perceptions of right and wrong [51]. In contrast to the focus of rational choice, norms are primarily non-consequentialist, i.e., they are concerned with actions rather than outcomes and with means rather than ends [52–54].

Although there appears to be little consistency on this in the social sciences literature, a distinction can be made between moral norms and social norms. According to Elster [53] social norms are shared norms which relate to appropriate conduct within a specific group of people and which are at least partly sustained by the approval and disapproval of others, while moral norms are personal norms concerning ethical values relating principally to the rights of other people in general and are largely independent of extrinsic influence. A key concept in sociology and social psychology is that moral norms are obligations that have become internalised so that they influence behaviour even in the absence of external pressures [55]. The extent to which social norms may be regarded as internalised, however, is not always clear from the literature [53, 56].

Norms, particularly norms of fairness and cooperation, may depend critically on observance by a sufficient number of others, or they may break down [52]. Norms may be undermined by external coercion, even though the norm may have encouraged the very behaviour desired by the external authority [57, 58]. Individuals may also personally evade the influence of norms in various ways.

Social influences are not always considered as normative. The behaviour and attitudes of others may be influential simply through close relationships between individuals in

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4 For example, Fishbein and Ajzen [107], Ajzen [108], Cialdini [109], also Grasmick and Green [110], Grasmick and Bursick [111].
5 See also Fehr and Falk [112].
6 Etzioni [54] suggests that a moral act should meet four criteria, each of which is necessary but not sufficient. First, it should reflect an imperative or feeling of obligation; second, it should be capable of being generalised to a variety of situations, i.e., it should reflect a principle; third, it should reflect symmetry in that no arbitrary discrimination is made between its objects; fourth, a moral act should affirm a commitment in the sense that the motivation is intrinsic.
7 Posner [113] suggests that there are four types of incentives to obey social norms: (i) private benefits, so that norms are self-enforcing; (ii) emotional, or passionate, responses; (iii) disapproval or ostracism; and (iv) feelings of guilt or shame (associated with internalised norms).
a group and through identification with the group. Granovetter [59] emphasises the importance of social relations (‘embeddedness’) in addition to what he calls ‘generalised morality’ (i.e., norms) in determining cooperation and trust within groups. Aronson [60] categorises responses to social influences in terms of “compliance, identification or internalisation”. In other words, individuals may act in response to the threat of formal or informal sanctions, or the example provided by significant others, or internalised norms. Sugden [61] views ‘conventions’ as the precursors of social norms that may become internalised through the human desire to obtain approval and avoid disapproval. Young [62] identifies six types of incentives that may act to determine compliance behaviour. These he lists as self-interest, enforcement, inducement, social pressure, obligation, and habit or practice, the last two of which would appear to include moral and social norms in the sense discussed above.

In a classic empirical study, the importance of norms in determining compliance with the law was investigated by Tyler [56], with particular attention to the (at least partly) internalised obligation that is derived from the legitimacy accorded to an external authority. Modern views of legitimacy in the social sciences derive from Weber [63] and suggest that acceptance of the legitimacy of an authority will encourage compliance with its laws even where those laws conflict with an individuals’ own self-interest [64]. In other words, legitimacy represents a perceived obligation to obey that is necessarily linked to political authority and is distinct from the influence of moral norms (indeed personal morality and legitimacy may conflict). The separation between legitimacy, morality and self-interest is not straightforward, however, nor is legitimacy a singular or absolute concept. To the extent that legitimacy is enduring it may approach the normative status of morality, for example, whereas legitimacy judged contemporaneously in terms of outcome may be said simply to reflect self-interest [56].

In the literature on local management or ‘co-management’ approaches to fisheries governance, it is often suggested that greater involvement of fishers in the management process will lead to increased levels of compliance because regulations will then be accorded greater legitimacy as a result [5, 6, 8, 9].

In the non-economic social sciences, even in the context of ‘commons’ problems like the fishery problems of compliance tend to focus away from deterrence, often to the point of ignoring economic forces altogether. But fishers seek economic returns; they have fixed factor costs, labour opportunity costs, etc., and hence will necessarily be sensitive to expected financial gains and penalties. On the other hand, even in fisheries enforcement studies based on the rational economic model, the importance of non-pecuniary factors is recognised. For example, Anderson and Lee [40] noted that “there are many issues

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8 For example, Schwartz [114] lists countervailing motives which can lead to the deactivation of a moral norm through ‘defensive redefinitions’. Assuming that an agent is responsible for his actions and that he has a choice of alternative actions, these may be characterised as (a) denial of consequence: the agent recognises the consequence of his action but denies to himself that this will follow in this case; (b) denial of responsibility: the agent changes his perception of his freedom to act; and (c) norm transformation: the agent changes his view of the moral norm which is applicable in the particular situation. Elster [52] considers how social norms and their influence can be manipulated.

9 These types of collective action/resource allocation problems are generally referred to as ‘social dilemmas’ in the sociology/social psychology literature [115].
Hatcher and Pascoe

of moral values and ethics involved. First, all fishers may not cheat even though it is financially profitable to do so. Also, the suggestion that policies be implemented assuming that people will not comply with them has the potential for eroding social capital which depends on respect for the law.” Sutinen et al. [65] recognised that conventional economic models frequently fail to explain actual patterns of compliance seen in fisheries. In practice the high costs of enforcing regulations usually result in relatively low probabilities of detection, while at the same time penalties are not usually high enough to produce a strong deterrent effect. Nevertheless, these authors observed, in many cases a significant proportion of fishers do comply with regulations.

Despite the strict rationality assumptions of the standard economic model, the influence of non-pecuniary or non-instrumental factors such as behavioural norms has been incorporated, for example, into theoretical economic models of tax compliance [66–68], as well as models of compliance with fishery regulations [69–71].

There does exist a considerable and growing literature within economics on the desirability and possibility of including norms in the economic model of preferences. The literature on norms and the economic theory of utility and preferences has been extensively reviewed by Goldfarb and Griffith [72]. Griffith and Goldfarb [73], Hausman and McPherson [74] and Dowell et al. [75], with particular reference to ‘moral norms’. These authors distinguish between models which incorporate norms and those which model concern for others by including the income or utility of others in the own utility function (‘interdependent utility’).10 Dowell et al. [75] reject multiple or hierarchical utility models (see, for example [54]) in favour of a set of models in which moral or immoral behaviour results in a shift in utility from goods associated with that behaviour. As the authors note, these models are formally similar to the concept of ‘state-dependent’ utility in the literature on risk and uncertainty [43]. One version of their model allows for continuous rather than discontinuous effects on utility of behaviour that may be perceived as neither ‘completely moral’ or ‘completely immoral’. Using similar notation to theirs, we have, for example,

\[
U(X;I) = A(I) f(X),
\]

where \(X\) is a vector of consumption goods, \(I\) is some index of immoral behaviour and \(A(I)\) is a function that shifts the underlying utility function \(f(X)\). \(A\) takes values between zero and one depending on the ‘degree of morality’ of the behaviour. Small values of \(I\) may have little or no effect on \(f(X)\)—\(A\) remains at or near a value of 1—while at some point \(A\) may fall towards 0 so that utility is reduced to almost nothing for a very immoral act. The shape of \(A(I)\) in \(I\) will therefore reflect the type of behaviour and the particular moral norm under consideration. Hatcher and Gordon [69] adapt this model to modify an expected utility of violation expression similar to Eq. (14.1) above in their empirical study of compliance with fishery regulations.

To see the effect on a continuous compliance problem of the form in expression Eq. (14.2), let \(EU[\pi(x)]\) be the expected utility of net profit from choosing a particular

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10 For example, Hochman and Rodgers [116]; Skonhoft and Solstad [117] employ a similar approach to build a theoretical model of a fishery when fishers are altruistic.
level of $x$, i.e., the profit net of the expected penalty. For simplicity we can also standardise the legal limit of $x$ to 0, so that any $x > 0$ reflects a degree of non-compliance. Then we could write

$$EV[\pi(x)] = M(x) \cdot EU[\pi(x)],$$

(14.6)

where $M(x)$ represents the influence of a moral judgement on the choice of the level of $x$. By the product rule, the necessary first order condition for an optimal solution to Eq. (14.6) is

$$dM(x^*)/dx \cdot EU[\pi(x^*)] + M(x^*) \cdot dEU[\pi(x^*)]/dx = 0.$$  

(14.7)

Consider a situation in which, over the relevant range, no feasible choice of $x$ is regarded as in any way immoral. Then for any $x$ we have $M(x) = 1$ and $dM(x)/dx = 0$ so that Eq. (14.7) becomes

$$dEU[\pi(x^*)]/dx = 0,$$

(14.8)

which is the same as the decision rule for a strictly rational agent. If any $x > 0$ is regarded as so immoral that it is not chosen, then we have $x = 0$ by assumption, in which case there is no solution to Eq. (14.6). Otherwise, $x$ will be increased until condition Eq. (14.7) holds, at which point we must have

$$dM(x^*)/dx \cdot EU[\pi(x^*)] = - M(x^*) \cdot dEU[\pi(x^*)]/dx < 0.$$  

(14.9)

Here $0 < M(x^*) < 1$ and $0 < EV[\pi(x^*)] < EU[\pi(x^*)]$. The same basic approach could be used to model the effect of any normative judgement, or indeed the subjective effect of a more instrumental social influence, pressure to conform with the behaviour of one’s peers for example. Sutinen and Kuperan [71] proposed essentially the same type of effect for normative influences in a time allocation model of fishers’ behaviour, although they were not specific about the structural form.11 If we assume that the shape of the function $M(x)$ is determined by some underlying moral (normative) parameter $m$ so that for any $x$ we have $dM/dm < 0$, then by the Envelope Theorem $\partial EU(x)/\partial m < 0$ and hence $\partial x/\partial m < 0$. Thus the degree of violation is negatively related to the strength of moral judgement ($m$) against violating.

### 14.4 EMPIRICAL STUDIES OF COMPLIANCE IN FISHERIES

Relatively few published studies have modelled empirically the determinants of regulatory compliance in fisheries. Early empirical studies of individuals’ compliance with fishery regulations, which attempted to estimate violation functions based on the standard

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11 Becker and Murphy [118] use a similar approach to model social influences.
deterrence model, were reported by Sutinen and Gauvin [76] and Furlong [77]. Both these studies recognised factors other than those directly related to the monetary costs and benefits of violation in the determination of compliance behaviour, but their influence was not explored.12

Furlong [77] included in his theoretical compliance model a vector of variables to capture “personal and household” characteristics. In his estimation of the model using data from a survey of Canadian fishers he included variables for age, the proportion of the family currently unemployed and the proportion of family income derived from fishing. These were designed to serve as proxies for individual differences in “attitudes and proclivities” towards violation. In the model estimations presented these variables have the (intuitively) expected signs but only the age of the fisher is significant. Sutinen and Gauvin [76], in their study of compliance in the inshore lobster fishery of Massachusetts, similarly hypothesised that the incentive to violate is influenced by personal characteristics such as age, years in the fishery and the extent of fishery income dependence. In their estimated model these variables were all significant.

Sutinen and Kuperan [71] developed an extended compliance model that included, alongside monetary incentives and deterrence variables, variables relating to social influence, moral norms and the perceived legitimacy of the regulator and the regulations. Kuperan and Sutinen [78, 79] estimated a similar supply of violations model using data from a survey of fishers in Malaysia and their compliance with fishery zoning regulations. They found certain social, moral and legitimacy variables to be significant in determining levels of compliance.

Hatcher et al. [70] estimated an empirical model of compliance with quota restrictions among fishers in the UK. Significant explanatory variables for compliance in their model were the perceived risk of detection and the size of the expected fine if detected, but also the feeling of involvement in the design and implementation of regulations, an indicator of a norm of compliance and the perceived attitude of others. Gezelius [80] found that compliance by Norwegian fishers was associated with an informally enforced (i.e., by other fishers) set of norms. These norms themselves permitted violation of some regulations that were not considered legitimate. Social factors were found to be major influences on compliance in Italian fisheries. Gambino et al. [81] found that violation behaviour of the Italian fishers interviewed was mainly affected by (in order of importance) social pressure, their judgement about legitimacy, moral influence and their judgement about the effectiveness of the enforcement system.

In a study of Danish fishers [82], economic benefits from non-compliance were demonstrated to be the most important single factor influencing compliance of the fishers. However, norms, inclusion in the decision making process (affecting legitimacy), beliefs about the behaviour of others and belief (or disbelief) that the system would provide conservation benefits were all found to be contributing factors.

A more recent UK study by Hatcher and Gordon [69] collected more detailed data on fishers’ perceptions and experience of enforcement and, in addition, sought to measure the financial incentive to cheat. A more sensitive econometric model was used which enabled

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12 In such studies, any differences in violation behaviour not attributable to measured variables are, implicitly, subsumed within the error term.
the influences on degrees of violation to be investigated. Although the authors still found some evidence of normative influences on quota violation levels, their results suggested that “conventional” economic incentives predominated. Interestingly, fishers in this survey sample appeared not to form their subjective risk of getting caught on the basis of their past experience of either landings inspections or logbook inspections at sea. It appeared that frequency of inspections per se had little effect on the perceived chance of detection, perhaps because violators were skilled at concealing over-quota fish from inspectors, or because the standard of inspections was simply inadequate. There was, however, a strong association between the subjective risk of detection and past experience of successful convictions. Levels of quota violations in the fishery appeared to be driven mainly by financial incentives. Higher violations were positively associated with perceptions of greater constraints on earnings and negatively associated with the perceived risk of getting caught. No social influence appeared to be significant in the determination of violation levels in this study, but there was some evidence of a normative influence on violations, with the recognition of the conservation value of quotas being more associated with lower levels of violation (although it was acknowledged that violators might seek to justify their actions by negating the utility of quota controls in interview). A significant association was also found between recognition of the authority of Producers’ Organisations in implementing quota controls and lower levels of violation, and this influence appeared to be more important than recognition of the effectiveness of quotas.

14.5 THE RELATIONSHIP BETWEEN NON-COMPLIANCE AND THE MANAGEMENT REGIME

There is a clear relationship between the potential for non-compliance and the management regime under which fishers operate. For example, it is only possible to land over-quota catches if there is a quota to begin with. Similarly, landing undersized fish requires there to be a minimum landing size restriction.

With input controls, different incentives are created to circumvent the management regulations. For example, restrictions on days fished can be circumvented to some extent through increasing the number of days fished, fishing for longer each day, employing larger boats, etc. This input substitution, while still formally complying with the regulation, results in greater than expected catch levels. Thus, while complying with the letter of the regulation, it is not complying with the spirit of the regulation.

There is some evidence, from the empirical studies previously cited, that compliance can be affected by the way management is imposed as much as the regulations themselves. In particular, compliance may be improved through industry involvement in the management system. That is, a system of co-management as discussed in Section 14.2. As noted previously, regulations designed and implemented with industry involvement may be viewed as having greater legitimacy than regulations that are imposed with no industry involvement.

A number of empirical studies found that economic incentives played a major role in the compliance behaviour of fishers. Consequently, reducing the incentives to cheat by, for example, increasing enforcement effort (i.e., increasing the probability of detection)
and increased penalties, should result in improved compliance. However, even if the relationship between increased enforcement and compliance were linear (so that doubling enforcement effort doubled compliance) the costs of such an increase in enforcement would be substantial and might outweigh the costs of non-compliance. Increasing the penalty associated with being caught undertaking illegal activity is likely to be more cost-effective, but would require a substantial change in most judicial systems, something that is often beyond the control of fisheries managers.

The empirical studies cited previously were primarily concerned with economic incentives to cheat (i.e., the benefits of non-compliance versus the expected cost of prosecution). However, there is evidence that economic incentives to comply (rather than not comply) can also be generated when some form of property right is introduced into the management regime. For example, quota systems, and individual transferable quota (ITQ) systems in particular, have often been criticised for the incentives they provide to discard (compliance) and land illegally (non-compliance) [83]. However, there is evidence of increased resource stewardship by the fishing industry, for example under the New Zealand ITQ system. Here the industry carries out considerable biological research on its own and regularly confers with the Ministry of Fisheries in setting TACs. The industry is also active in the enforcement of fisheries rules, and in some cases takes on full management responsibility (e.g., in the southern scallop fishery, [84–88].

The development of self-management in the New Zealand can be viewed as a direct consequence of the secure property rights conferred by the ITQ system [89]. ITQs provide a means for resolving conflicts over the distribution of the resource, and are therefore considered key to the development and success of the co-management structure evolving in New Zealand [86]. However, Dubbink and van Vliet (2000) suggested that even an ITQ system on its own is not effective without some form of contemporaneous stakeholder management. In the NZ case, concerns about poor enforcement and the scientific processes underlying quota setting led to the development of greater stakeholder participation in the management of the fisheries. As a consequence of the improved property rights inherent in an ITQ system, individual fishers expected to gain directly from an improved resource, and were therefore willing to co-operate rather than compete.

Sutinen [90] found that countries that utilised rights based management approaches tended to also move towards a system of co-management. These countries have found that the co-management approach reduces administrative costs and greatly improves compliance with management regulations. In contrast, only fisheries with relatively few participants were able to successfully develop co-management structures in the absence of rights-based management systems.

14.6 MODELLING NON-COMPLIANCE

The focus in the empirical studies in Section 14.4 was on the factors contributing to the decision to comply or not comply. From Sections 14.4 and 14.5 it is apparent that the incentives generated through the management regime (including the benefits and expected costs of non-compliance), perceptions of legitimacy, the compliance behaviour of others, and so on, will all affect the individual fisher’s compliance decisions. In most cases, the
outcomes of non-compliance have not been directly investigated, although in some studies estimates of illegal landings have been made. For example, Hatcher et al. [70] found that 43.5% of fishers interviewed estimated that their landings had been over-quota by 10% or less and 29% said their landings had been over-quota by 25% or more. Svelle et al. [91] estimated that discards and illegal landings of cod in the North Sea accounted for 22% of the total catch weight and 51% of the number of caught fish. However, a distinction was not made between illegal landings (non-compliance) and discarding (compliance with adverse effects).

A feature of bioeconomic models of fisheries and other resource-based industries to date has been that compliance is implicitly assumed. For example, quotas are modelled as binding functions on landings (if not catch). In some models [92, 93], discarding has been explicitly modelled as compliance behaviour. That is, overquota catch estimated in the model is all assumed to be discarded. This assumption of compliance has resulted in some potential distortions in the model results. For example, in a model of the Dutch beam trawling fleet [94], it was assumed that fishing would cease as soon as quota of one of the several species was achieved, even if quota on the other species was available. This resulted in an underestimate of the overall catch and economic benefits from the fishery, as fishers would have continued to fish and discard overquota catch (again, assuming compliance behaviour) provided the value of the catch that could be landed exceeded the additional cost of catching it.

Where non-compliance has been explicitly considered in fisheries models, it has generally involved a comparison of compliance with restrictions versus the case with no restrictions (i.e., effectively zero compliance). For example, Hutton et al. [95] modelled the benefits of compliance versus non-compliance with various minimum size and effort restrictions for a South African fishery using a bioeconomic simulation model. Non-compliance was modelled through removing the restrictions imposed in the compliance simulation.14

Such an approach effectively represents the two extremes of compliance: perfect compliance and perfect non-compliance. However, the studies of compliance in fisheries have found that compliance generally falls somewhere between these extremes, with part of the fishery complying and other parts not complying, or complying to varying degrees. As a result, the “true” outcome from a management strategy will fall somewhere between the two extremes that can be readily modelled. Kritzer [96] attempted to overcome this through assuming, in the case of marine protected areas (MPAs), that some of the fishing effort that should be displaced through the designation of the MPA would remain within the protected area (i.e., imperfect compliance). The effects of varying amounts of illegal fishing activity were examined.

13 The model was actually based on cooperation and non-cooperation rather than compliance per se, but cooperation was defined in terms of compliance with the regulations and non-cooperation was defined in terms of non-compliance with the regulations.

14 Similar approaches are used in other fields also. For example, Pacini et al. [119] estimated the costs of compliance with EU agricultural pollution regulations by comparing a perfect compliance scenario with a perfect non-compliance (i.e., modelled as no regulation) scenario.

15 “True” in this context is only relative, as models contain many other simplifications and assumptions that may result in a divergence between the modelled and real-world impact.
Modelling studies in other fields have attempted to incorporate the effects of non-compliance through assuming compliance with higher levels than the regulation permits. For example, Meyburg et al. [97] modelled the economic impacts of weight limits for heavy vehicles by assuming perfect compliance with a limit above that actually imposed (i.e., non-compliance with the actual regulation). Similarly, Sadiq et al. [98] modelled wastewater treatment effectiveness by assuming an amount of agricultural pollution above that that the regulations prescribed. In both examples, the determination of the ‘excess’ above the regulated restriction (i.e., the effective compliance limit) was relatively subjective, based largely on anecdotal evidence.

In theory, it should be possible to determine the effective compliance limits based on the likelihood of compliance. The previous studies of compliance in fisheries effectively determined the probability of compliance based on the characteristics of the fishery, the fishers and the management regime. Given this, the effective compliance limit could be estimated as the expected outcome given the probability of compliance. That is, average of the limit multiplied by the proportion expected to comply and the unconstrained outcome multiplied by the proportion expected not to comply (for example, the capacity output of the fleet in the case of quotas or the potential effort in the case of effort controls). This could be further refined where information on the expected extent of violation is also available (e.g., in the study by Hatcher and Gordon, [69]). To date, such an approach has not been applied to any fisheries bioeconomic model, nor to any model of any other sector.

14.7 CONCLUSIONS

Non-compliance affects, and is affected by, the management system that is imposed on the fishery. As a result, it distorts the knowledge base upon which management is formulated. In particular, it distorts the data that are subsequently used in fisheries analysis, and reduces the confidence of the stakeholders in the quality of the subsequent policy advice. This, in turn, may lead to even lower levels of compliance, with even greater impacts on the knowledge base. Understanding why fishers do not comply with regulations is therefore essential in developing appropriate fisheries management systems, and is hence an important part of the fisheries knowledge base itself.

Empirical analysis of individual compliance behaviour in fisheries suggests that economic incentives are a major factor determining the level of compliance. These incentives work in two different directions. Incentives to not comply relate to the benefits of non-compliance and the expected costs of non-compliance (a function of the probability of detection and the expected penalty). Changing these incentives through either increasing enforcement or increasing the penalty associated with prosecuting illegal activity is not necessarily possible or desirable. In the case of enforcement, the additional costs may outweigh the benefits associated with the reduction in illegal activity, while in the case of the penalty, this may be largely beyond the control of the fishery manager.

Incentives to comply, however, might be improved through increased use of rights-based management and also increased industry involvement, or ‘co-management’. These are, in practice, largely complementary activities as the use of rights-based management
Non-compliance with fishery regulations has generally led to the development of co-management structures. The benefits of improved compliance are captured directly by the individual fishers through the improved resource base as a result of their strengthened property rights, while the co-management structures arguably increase the legitimacy of the management measures.

Incorporating the effects of non-compliance in fisheries bioeconomic models in order to estimate more realistic outcomes of management policies is less straightforward. Most studies have either assumed a dichotomous set of scenarios (i.e., perfect compliance or perfect non-compliance), or have assumed an effective compliance limit based on a level of restriction that is likely to be complied with. This, for example, may be a total allowable catch that is 10–20% higher than that imposed. These effective compliance limits have generally been subjectively determined. A potential approach that has not yet been applied to fisheries (nor to other industries) is to use the probability of compliance information in conjunction with the dichotomous scenarios in order to objectively define effective compliance limits. This could allow these limits to be endogenously determined with changes in management systems.

Non-compliance affects the knowledge base through distorting the information used in fisheries stock and economic assessments. As a consequence, the controls imposed may not fully reflect the true underlying conditions in the fishery. Ironically, the existence of these distortions in the knowledge base reduces fishers’ perception of the legitimacy of the controls, further reducing the probability of compliance.

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Section 3

SCENARIO MODELLING AS SUPPORT FOR FISHERIES MANAGEMENT SYSTEM EVALUATION
Glossary of concepts used within chapters 15 and 16

**Management Strategy** refers to the combination of specific management objectives and associate implementation measures.

**Management Strategy Evaluation** (MSE) consists of simulations of some or all elements of the management strategy, complemented with analyzes of those elements of the strategy which or not amendable to quantitative analysis.

**Management Procedure** (MP) is a simulation-tested set of rules used to determine management actions, in which the data, assessment methods and the harvest control rules for implementing management actions (i.e., the rules used for decision making) are pre-specified.

**Operational Management Procedure** (OMP) is defined as a management procedure that is currently being used to determine management actions or has been intensively tested by a competent management body to a level where it could be used in practice.

**Management Procedure Evaluation Framework** (MPEF) provides a simulation tool that allows OMPs to be developed in a manner that meets the requirements of FAO’s precautionary approach to fisheries management.
Chapter 15

Operational Management Procedures: An Introduction to the Use of Evaluation Frameworks

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15.1 INTRODUCTION

This chapter describes various aspects of management procedures, including the philosophy behind them, how they are constructed based on available data and knowledge, their objectives, the methods used to evaluate them, and their differences to and advantages over traditional stock assessment and management regimes. In addition, it also reviews some examples of management procedures.

An important benefit of management procedures is they can be designed to satisfy a variety of pre-agreed management objectives, including both biological and economic factors, making them more likely to be acceptable to a wide range of user groups. However, the modelling philosophy may be difficult to understand for non-experts, which means that extra effort is needed to explain them and how they are designed.

A management procedure is a simulation-tested set of rules used to determine management actions, in which the data, assessment methods and the harvest control rules for implementing management actions (i.e., the rules used for decision making) are pre-specified [1]. An operational management procedure (OMP)¹ is defined as a management procedure that is currently being used to determine management actions or has been intensively tested by a competent management body to a level where it could be used in practice.

¹The original intention of introducing the qualifier “operational” was to distinguish between management procedures that could be put into operation because of the availability of data and formulae to do so, from purely conceptual ideas.
Management procedures are rigorously tested before implementation using computer simulation to ensure robustness to a wide range of uncertainties. This is done through the development of an operating model that represents the underlying reality against which candidate management procedures are tested with respect to explicitly stated and prioritised management objectives (e.g., low-risk of depletion, high and stable catches). The approach was pioneered by the Scientific Committee of the International Whaling Commission [2–8], and is also being used in fisheries management, particularly in South Africa [1, 9–13] and Australia [14–18]. Management procedures have now been implemented in a variety of regions to meet a range of social objectives and legislative requirements. For example, the management procedure approach is becoming more widespread in Europe because the current fisheries policy in the European Union [19] recommends that multi-annual recovery plans should be used for depleted stocks. Table 15.1 provides examples of management procedures, only some of which have been implemented in practice.

Table 15.1  List of operational management procedures

<table>
<thead>
<tr>
<th>Country or Management Agency</th>
<th>Primary References</th>
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<tr>
<td>1.2 Brown and grooved tiger prawns</td>
<td>[15]</td>
</tr>
<tr>
<td>1.3 East coast tuna</td>
<td>[14]</td>
</tr>
<tr>
<td>1.4 Eastern stock of gemfish</td>
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</tr>
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<tr>
<td><strong>International Whaling Commission</strong></td>
<td></td>
</tr>
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<td>[2, 7, 8, 43]</td>
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<tr>
<td>3.2 The Bering-Chukchi-Beaufort Seas stock of bowhead whales</td>
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<tr>
<td>3.3 The Eastern North Pacific stock of gray whales</td>
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<td>4.1 Cape fur seals</td>
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</tr>
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<td>5.1 Rock lobster</td>
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<td><strong>Norway-Russia</strong></td>
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<td>6.1 Cod</td>
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<td>7.4 West coast rock lobster</td>
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<tr>
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<td></td>
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<tr>
<td>8.1 Pacific sardine</td>
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<td>8.2 Pacific mackerel</td>
<td>[78, 79]</td>
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<tr>
<td><strong>UK</strong></td>
<td></td>
</tr>
<tr>
<td>9.1 Blackwater herring</td>
<td>[39]</td>
</tr>
</tbody>
</table>
15.2 RATIONALE

The conventional process for providing scientific recommendations for fisheries management involves selecting the current ‘best’ assessment of stock status and using this as the basis for the regulatory mechanism to be applied. Stock assessments use increasingly complex models with significant input and modification by scientists, often on an annual basis, while the regulatory mechanisms include Total Allowable Catches (TACs) or limiting fishing effort. These mechanisms are often based on biological reference points, such as the maximum sustainable yield (MSY), the biomass associated with MSY ($B_{MSY}$) or the corresponding fishing-induced mortality ($F_{MSY}$, or a proxy such as $F_{max}$ or $F_{0.1}$). In the case of the scientific advice provided by ICES on behalf of the European Commission, the specification of TACs generally relies on defining precautionary reference points ($F_{pa}$ and $B_{pa}$, for fishing mortality and biomass, respectively) to ensure that the real state of the stock is on the safe side of the limit reference points ($F_{lim}$ and $B_{lim}$) with a high probability [20]. However, the differences between the precautionary reference points and the limit reference points are often arbitrary. OMPs offer one way to define these differences.

Currently, scientific advice and management is often done in a fairly ad hoc manner. This is because most biological reference points fail to fully take the uncertainty associated with the quantities used to calculate them into account, with potentially dire consequences for the fish stocks to which they apply [21]. For example, some fishing mortality-based reference points are known to lead to unsustainable exploitation rates for certain species [22, 23]. This view is supported by the simulation work conducted by Kell et al. [24, 25], which showed that ICES biomass and fishing mortality reference points are not always consistent, and are often clearly inappropriate. This is because, when deriving these reference points, important sources of uncertainty about the likely and alternative dynamics of the stocks and management system are ignored, for example, with respect to the ability to collect data, assess the stock, and implement management measures.

The FAO Technical Consultation on the Precautionary Approach to Capture Fisheries [26] recommended the use of decision (or harvest control) rules as one of the elements of the precautionary approach to fishery management. These rules should “specify in advance what actions should be taken when specified deviation from the operational targets and constraints are observed” and should be able to “respond to unexpected or unpredictable events with minimum delay”. Although harvest control rules based on biological reference points have been used widely to provide fisheries management advice, and may include several precautionary elements, it does not necessarily follow that they will be precautionary in practice [27]. This is because most harvest control rules are not evaluated formally to determine the extent to which they achieve the goals for which they were designed, given the uncertainty inherent in the system being managed (Punt, in press). Simulation is increasingly being used as a means to perform these evaluations [27–29].

One of the weaknesses of fisheries science is that it is almost always impossible to conduct large-scale experiments on fish stocks, a notable exception being Sainsbury et al. [30] who used large-scale experimentation to identify causes for ecosystem shifts on Australia’s North West Shelf. It is therefore almost always impossible to ascertain the
actual causal dependencies in a system. Although simulation testing of OMPs cannot replace the missing experiments, it offers a tool to evaluate the consequences of alternative hypotheses in terms of achieving management objectives. In addition, it helps focus discussion by stakeholders on the strategic rather than the tactical aspects of fisheries management, for example, deciding upon whether effort or catches should be controlled, rather than arguing about this year’s TAC level.

15.3 MANAGEMENT OBJECTIVES

It is necessary to identify and agree on explicitly stated and prioritised management objectives before evaluating alternative management procedures. These objectives will differ among management jurisdictions, countries and organizations, and can include biological, economic and social aspects. Objectives can be divided into high-level (policy, e.g., whether to favour conservation or short-term economic gain), and operational (specific, e.g., which stocks or fleets to regulate in a mixed fishery) objectives. It is often the case, as outlined below, that management objectives are expressed qualitatively. These qualitative objectives need to be made quantitative for them to form the basis for an evaluation of management procedures. This should be done, as far as possible, with the decision makers and stakeholders, prior to any evaluations. The results of the evaluations may, in fact, help to determine the extent to which the management objectives are in conflict, and hence whether different objectives imply different management strategies.

Examples of management objectives for some of the OMPs identified in Table 15.1 are given below.

The objectives of the IWC were established by the International Convention on the Regulation of Whaling (ICRW) and include the generic objectives: “... to establish a system of international regulation for the whale fisheries to ensure proper and effective conservation and development of whale stocks...” and “... make possible the orderly development of the whaling industry” [5].

Within a country, management objectives are generally set by legislation. For example, in Australia, the Fisheries Management Act No. 162 [31] states the management objectives to be:

- implementing efficient and cost effective management;
- exploitation of fisheries resources in accord with the principles of ecologically sustainable development;
- maximising economic efficiency in the exploitation of fisheries resources;
- accountability of fisheries management; and
- meeting cost recovery targets set by government.

In Iceland, management objectives are detailed in the Fisheries Management Act No. 38 [32], which defines them to be “to promote [the] conservation and efficient utilisation [of exploitable marine stocks] and thus to ensure stable employment and settlement throughout the country”.
In the United States of America, the Magnuson–Stevens Fishery Conservation and Management Act [33] makes the U.S. Federal government responsible for the management of fisheries from 3 to 200 miles off the coast for most species, and beyond 200 miles for anadromous species such as salmon. The objective of current management is to provide Americans with commercial and recreational fishing opportunities, and a safe supply of high quality seafood. This is achieved by aiming to:

- ensure biological and economic sustainability;
- maintain stock levels at biologically healthy levels and ensure optimal harvesting of fish over time, using the least-cost levels of capital, labour, and other resources; and
- allocate the harvest among user groups equitably.

In South Africa, fisheries management has clearly defined policy objectives aimed at achieving the ‘best possible use’ of living marine resources [34]. OMPs are being developed for all ‘significant’ marine resources in South Africa, and long-term management plans explicitly refer to them. These OMPs are based on scientific principles, recognizing the inherent variability of resources and the interdependence of the components of marine ecosystems. The management plans are being developed through a cooperative process involving all interested parties and include monitoring and control programmes, as well as enforcement of fishing regulations. They require consideration of the socio-economic implications of altered levels of utilization (e.g., the effect of a reduced TAC on employment).

Namibia’s primary objectives immediately following Independence in 1990 were to rebuild its overexploited and depleted resources to maximum sustainable levels, maintain its other exploited resources at healthy levels, and seek out opportunities for developing new fisheries [35]. Subsequently, objectives related to the conservation of stocks have included:

- maintaining/rebuilding marine resources to support long-term sustainable yields;
- taking full account of the ecosystem approach to fisheries;
- developing and implementing fishery management plans to determine reference points, management strategies and research priorities for the major commercial resources; and
- applying the precautionary approach to ensure exploitation at levels generally below maximum sustainable yields.

At the 31st meeting of the Joint Russian-Norwegian Fisheries Commission in November 2002 [36], Russia and Norway agreed that management strategies for artic cod and haddock should take into account:

- conditions for high long-term yield from the stocks;
- achievement of year-to-year stability in TACs; and
- full utilisation of all available information on stock development.

The approach in New Zealand has been to define fisheries management goals within an overall government strategic vision of “a clean, healthy and unique environment,
sustaining nature and people’s need and aspirations”, in collaboration with various interest
groups such as Maori, environmental groups, and recreational and commercial fishers
[37]. Goals for fisheries management include:

- maintaining the biological diversity of the aquatic environment;
- protecting habitats of particular significance for fisheries management;
- maintaining species at levels that ensure their long-term viability;
- ensuring the Crown delivers on its fisheries obligations to the Maori; and
- ensuring efficient use of resources, including reducing business compliance costs.

In the EU, the current legislation of the CFP [19] defines several objectives: “The
Common Fisheries Policy shall ensure exploitation of living resources that provides
sustainable economic, environmental and social conditions” and “For this purpose, the
Community shall apply the precautionary approach in taking measures designed to protect
and conserve living aquatic resources, to provide for the sustainable exploitation and to
minimise the impact of fishing activities on marine ecosystems. It shall aim at progressive
implementation of an ecosystem-based approach to fisheries management. It shall aim to
contribute to efficient fishing activities within an economically viable and competitive
fisheries and aquaculture industry, providing a fair standard of living for those who depend
on fishing activities and taking into account the interests of consumers”. In addition,
one of the basic principals of the Common Fisheries Policy (CFP) of the European
Union is relative stability (Articles 32–37 of the EC Treaty; [38]): “[The] principle of
relative stability, based in particular on historical catch levels, means maintaining a fixed
percentage of authorised fishing effort for the main commercial species for each Member
State. Fishing effort should be generally stable in the long term...”. Within mixed
fisheries (stocks and fleets) this may mean that the management objectives are conflicting
and hence difficult to achieve.

In the UK the Blackwater herring OMP was developed to ensure a sustainable and
profitable fishery and to reduce assessment and management costs [39].

The above examples illustrate the wide range of possible management objectives. One consequence of this wide range, and its largely qualitative nature, is that managers
often experience great difficulty in determining how their objectives can be expressed
quantitatively [27, 40]. Consequently, representing management objectives quantitatively
is often one of the most difficult tasks to accomplish when evaluating OMPs [21].

McDonald et al. [41] provide an example of a quantitative interpretation of a broadly
stated objective. In their example, the objective “to maximise the value of the harvest,
without threatening the sustainability of the fishery” is interpreted as “to allow the greatest
dollar value of catch to be taken over the long term, so long as the breeding stock left in
the water is likely to be at least 30% of its original level”.

Unfortunately, the inability to quantify management objectives has meant that OMP
evaluations have not always been based on the full range of management objectives. For example, Punt et al. [42] were only able to take explicit account of two of the five
objectives for Australian fisheries in their evaluation of OMPs for school and gummy
shark off southern Australia. Specifically, they were not able to relate management
practices to cost recovery, accountability and cost efficiency, and could not fully account
for the impact of the choice of an OMP on the full range of ecosystem services.
Scientists commonly refer to the problem of poorly defined management objectives making it difficult to recommend a single management action. However, at a minimum, it should be still possible to rank the alternative OMPs relative to each management objective, so that stakeholders can evaluate the linkage between objectives and alternative OMPs. This will support the “broad involvement of stakeholders”, as requested in, for example, the CFP legislation, and is consistent with good practice within any democratic process.

15.4 DESCRIPTION OF OMP APPROACH

A Management Procedure Evaluation Framework (MPEF; Fig. 15.1) provides a simulation tool that allows OMPs to be developed in a manner that meets the requirements of FAO’s precautionary approach to fisheries management [26]. A MPEF was first used by the

Fig. 15.1. A conceptual representation of the Management Procedure Evaluation Framework (after Kell et al. [25]; reproduced with permission of the ICES Journal of Marine Science).
In the late 1980s and early 1990s, OMPs were developed as a tool to apply OMPs for commercial whaling of baleen whales [78, 43]. As noted above, OMPs translate data from the fishery into a regulatory mechanism (e.g., a TAC or maximum fishing effort), either by fitting an assessment model to observed data (model-based approach), or by using a more empirical method, such as following trends in a survey-based index of abundance (model-free approach; [29]). These rules are tested within a MPEF to ensure that they are reasonably robust in terms of expected catches and risk to the resource, given the prevailing uncertainties about resource status and dynamics.

The rules should be agreed upon by all parties concerned (scientists, industry, managers, and other stakeholders) before being implemented. Once implemented, it is desirable that OMPs be left to operate ‘automatically’ for a set period (e.g., 3–5 years, in the case of South African fish stocks) where scientists should not seek to alter the recommendations that they provide unless very strong evidence pointing to such a need becomes available. After this period, the OMPs should be reviewed and modified as necessary in the light of any changes in understanding of the resource or fishery that may have occurred in the interim [1, 12].

An important source of uncertainty when managing a fish stock can be the behaviour of institutions (managers), scientists, and fishermen, which is often difficult to quantify. However, the MPEF attempts to reduce this uncertainty by ensuring that all parties are involved in specifying and agreeing the management objectives and the candidate OMPs. In addition, account can be taken in the MPEF of hypotheses for the possible response of fishers and managers to scientific management recommendations, including infringement behaviours [44] and ignoring (or modifying) such recommendations [15]. ‘Implementation uncertainty’ can be a major source of uncertainty for some cases (e.g., tiger prawns in Australia’s Northern Prawn Fishery; [15]).

An MPEF includes (in addition to the OMP itself) an Operating Model, which represents the ‘true’ dynamics of the system, against which performance is measured, an Observation Error Model, which generates the types of data available for use by the OMP in a realistic manner, and Performance Statistics which are used to evaluate the performance of the OMP against the management objectives (Fig. 15.1; [25]). The Observation Error Model and OMP together reflect the ‘perceived’ system (i.e., our perception of the system following the collection of observations and the assessment of stock status), while the ‘true’ system (i.e., the combination of the Operating Model and the Starting State of the system) coupled with the Observation Error Model captures plausible hypotheses about the system dynamics. The Observation Error Model reflects the transition between the ‘true’ and ‘perceived’ systems.

The OMP approach involves determining how well it is possible to control exploitation, given information from the Observation Error Model about the (unknown) future. Care should be taken when selecting how the observations are generated. For example, if the operating model and the assessment model underlying the OMPs make the same statistical assumptions about some data source (e.g., that it is lognormally distributed) this could lead to an unrealistically optimistic appraisal of the ability to conduct assessments. This could, in turn, lead to the selection of an OMP that does not perform as well as anticipated. A possible solution to this problem would be to have ‘blind testing’, where the operating model is developed by different scientists to those who develop the candidate OMPs.
The basic MPEF can be extended. For example, McAllister et al. [29] identify the need for a model to capture managers possibly not implementing OMP-derived TACs or other regulatory measures because of factors not incorporated when calculating these measures, such as socio-economic factors. McAllister et al. [29] also explicitly identify the need for an ‘implementation model’, which includes converting TACs into catches, and takes into account factors that may cause these two quantities to differ, such as limitations imposed by bycatch considerations. Another extension is to incorporate fleet adaptation, the learning processes of fishers, and economic considerations, to better reflect the behaviour of the fisheries system being modelled [44–46].

The OMP approach need not have as its end-product an agreed OMP. For example, in Australia, the OMP approach is used to guide (rather than prescribe) short- and long-term decision-making [47] and to select appropriate methods of stock assessment. Furthermore, the studies of Kell et al. [24, 25, 48] focussed on evaluating restrictions in inter-annual variations for TACs on behalf of the European Commission.

15.5 MODELLING CONSIDERATIONS

15.5.1 Modelling uncertainty

The MPEF aims to evaluate the extent to which candidate management procedures are robust to uncertainty. As a result, the extent to which the MPEF provides an adequate evaluation of candidate management procedures depends on the extent to which the key sources of uncertainty are captured. The range of uncertainties considered during management procedure evaluations can include the following [24, 25, 49, 50]:

- process error—natural variation in dynamic processes such as recruitment, somatic growth, natural mortality, and the selectivity of the fishery;
- observation error—related to collecting observations from a system (e.g., age sampling, catches, surveys);
- estimation error—related to estimating parameters, both in the operating model, and, if a model-based approach is used, in the assessment model within the OMP that leads to the perception of current stock status;
- model error—related to uncertainty about model structure (e.g., causal assumptions of the models), both in the operating model and in the OMP; the operating model needs to be sufficiently complicated to capture all key biological processes, and several alternative operating models may be developed, while if the OMP includes an assessment model (almost always simpler than the operating model), it will never capture the true complexity of the system dynamics; and
- implementation error—because management actions are never implemented perfectly and may result in realised catches that differ from those intended.

Figure 15.2 provides an example of a single simulation run from an MPEF, highlighting the difference between the true and perceived systems.
Kell et al. [24, 25] showed that when realistic sources and levels of uncertainty are considered, far from optimal management outcomes may result when current management approaches are used. For example, the current ICES biomass and fishing mortality reference points are not always consistent, and several are clearly inappropriate. This is because the types of projection used by ICES do not incorporate important lags between assessing stock status and implementing management measures, and they also ignore important sources of uncertainty about the dynamics of the system, as well as the ability to collect data and implement management regulations (i.e., model, observation, and implementation error respectively). Kell et al. [24, 25] also showed that better management is not necessarily going to be achieved by improving stock assessments, because, even with a perfect assessment (where the simulated working group knew stock status perfectly), stocks may collapse at fishing levels that standard stochastic projections would suggest are safe.

15.5.2 Constructing operating models

Operating models are simulation models that represent the underlying situation in the fishery (e.g., stock dynamics, fishers’ behaviour) and that capture the existing knowledge and data for the fishery, including both what is known and what is not. A set of structurally
different operating models is needed to evaluate the robustness of candidate OMPs against the full range of uncertainty that applies to the fishery under consideration. Peterman [51] compared operating models to flight simulators: “the latter include detailed dynamic feedback processes to help pilots determine which decision-making protocols are best in the presence of a wide range of possible, but uncertain, simulated contingencies”.

Operating model components used in the MPEF, whether biological, economic or bio-economic, must be 'conditioned' on the available data. A model is conditioned on data if it is fitted to the data so that the model predictions of the data are approximately consistent with the actual data. This conditioning process can lead to an undesirably narrow range of scenarios, with the result that candidate OMPs are only tested against scenarios which have either been observed, or are at least fairly likely given the observed data. This assumption is well justified for the base case (or most likely) scenario.

However, conditioning need not lead to an unnecessarily narrow range of scenarios, and OMPs should be tested for problematic cases which have not yet been observed, but which are nevertheless possible, i.e., representing ‘justified concerns’ to which the OMP should be robust. For example, there was an interest to examine the consequences of productivity being lower than believed when developing the OMP for the Bering–Chukchi–Beaufort Seas stock of bowhead whales (for which the estimates of abundance have been increasing rapidly in recent years [52]). As a result, an operating model was developed in which productivity was low, but there has been a change in the bias of the surveys (so that the operating model mimics a rapid increase in survey estimates by changes in survey bias rather than high productivity).

Joint probability distributions for the values for the parameters of the operating model should be obtained during conditioning to represent estimation uncertainty. Each simulation should then be based on a random draw from this distribution. Wherever possible, the credibility of each alternative operating model should be computed as part of the conditioning process by calculating the probability of the model given the data.

There will always be alternative aims when evaluating OMPs. The following represent four different ways to construct operating models. These considerations are expressed mostly in a Bayesian context, but there are other valid philosophies when constructing operating models. Many evaluations of OMPs use all four of these ways to some extent. The amount of knowledge, data requirements, and complexity of implementation differs quite markedly among these four types.

I The operating model is the currently-used stock assessment model. Although use of the assessment model as the operating model seems to imply that assessment models describe nature almost perfectly, if a OMP cannot perform well when reality is as simple as implied by an assessment model, it is unlikely to perform adequately for more realistic representations of uncertainty. Basing an operating model on the current assessment model has arguably the lowest demands for knowledge and data.

II The operating model is a model that can represent all of the available (and valid) data. The values for the parameters of the operating model are based only on the data for the fishery under consideration (i.e., in Bayesian models, priors would be non-informative, so that only data would ‘speak’). This approach is based on the idea that all relevant data sets are available and that only data matter when considering
future events. The operating model need not be identical to the models underlying the assessments used as part of the OMPs. This approach assumes that, with no information to the contrary, the future will be similar to the past, which is a strong assumption.

III As for II except that, in Bayesian models, priors would describe in a formal way the knowledge of scientists related to the validity of information sources. Probabilities, other than those available from data, may come from, for example, meta-analyses. This is still a data-orientated approach, but other data sources than those for the fishery under consideration have an impact when conditioning the operating model.

IV As for III except that the emphasis is on expert beliefs and other a priori information about the processes that may affect the behaviour of management systems in the future (i.e., the focus is on the future, not on fitting historical data). This is a less data-, and more hypothesis-orientated approach. For example, climatic change studies may show that a regime shift is possible (even though one has never been seen in the historical data sets) and should be taken into account when selecting ways to provide management advice. It is important therefore that operating models are flexible so that they can deal with such factors.

It may be considered, that IV is related to the testing the robustness of OMPs to alternative hypothesis about the dynamics of the system and that II represents the base (or ‘most likely’) case. As the aim of the MPEF framework is to test different types of plausible hypothesis in a management context, type III and IV models are perhaps the most important after a base case model is agreed. The OMP selected after the simulation evaluation is completed should be able to deal with most of the hypotheses that have some credibility, either in the light of the existing data or from other sources of information.

Attempts should be made to ensure consistency in assumptions when constructing, estimating parameters for, and projecting the operating model into the future. In other words, consistency in assumptions should be maintained both in the estimation of the parameters of the operating model using historical data and in the projections of the operating model into the future. For example, consistency should be maintained regarding the assumptions about the manner in which catches are misreported and the same error assumptions should be used when fitting the operating model and generating future data. Naturally, trials could explore the consequences of inconsistency between assumptions for the past and future if this is plausible (for example, high-grading may occur if future management is based on TACs in a fishery currently managed using input controls). This type of analysis may show how an improved management system, such as one with better commitment of stakeholders to management regulations, behaves compared to current system, helping to demonstrate the impact of humans.

Kuikka et al. [53] demonstrated how results from several operating models could be combined to evaluate the role of structural uncertainty, although they assumed that a single model applied over the entire time period considered. Under some circumstances it may be appropriate to incorporate several key forms of model uncertainty within a single operating model. For example, when dealing with uncertainty about the form of the stock-recruitment relationship, attempts could be made to formulate a single operating model that includes within it several functional forms for the stock-recruitment relationship (e.g.,
the Shepherd stock-recruitment relationship can capture the simpler Ricker, Beverton-Holt and Schaefer forms). After conditioning the operating model on the data, the simulation results can then either be marginalised for each alternative stock-recruitment relationship, or be averaged across all of them. Regime shifts are also good examples of where use of only one model type in the operating model would be misleading. The probability and length of, for example, periods of lower or higher productivity could be based on the results of other models, such as climatic models.

In instances in which there are insufficient data to estimate parameters for some hypothesised process in the operating model (e.g., historical misreporting of discards or future environmental conditions affecting recruitment), it may be necessary to specify a set of plausible values for the parameters. A set of operating models would then be constructed in which the operating model is fitted to the historical data pre-specifying the inestimable parameter to each of the plausible values while the other model parameters are estimated. The resulting distribution for the other model parameters would then be used together with the pre-specified values in the projections. Running projections for each alternative parameter value, and combining the results would account for the process by integrating over the range of plausible values for the inestimable parameters. For example Fromentin and Kell and Kell and Fromentin [54] evaluated MSY-based strategies for bluefin tuna using operating models that fitted the observed data, but were based on contrasting, but equally plausible, hypotheses about migration and recruitment dynamics. No prior probabilities were assigned to the alternative hypotheses because the aims of the study were to test the robustness of alternative assessment methods and management strategies to alternative plausible hypotheses about which there was little prior knowledge, and to identify where improved knowledge of stock dynamics was needed.

15.6 COMPARING THE OMP AND CONVENTIONAL APPROACHES

Even though OMPs have a different role from traditional stock assessment tools, it is useful to compare them. A key difference between the OMP and conventional approaches is that the latter requires a regular and time-consuming re-evaluation of data, assessment methodology and the process for setting the regulatory mechanism, as well as an (usually annual) update of the actual assessment results. Hilborn [55] forecast the end of the conventional stock assessment treadmill, where increasingly complex models are run each year, and may be significantly modified by scientists to produce an estimate of stock status that then determines management actions. Instead, he forecast that complex models would be used to describe alternative plausible states of nature against which simpler assessment/management models will be validated and tested before implementation, i.e., he forecast the use of the OMP approach, and implicitly recognised that it might not be possible to describe fully the true dynamic processes within any assessment model.

OMP{s require a more comprehensive re-evaluation, but at a less frequent interval. The interval between re-evaluations (typically only once every 3–5 years) depends on the dynamics of the fishery and also on the chance of an improved understanding of key biological and environmental processes. Use of OMPs arguably leads to a more time-efficient process, and a better basis to pursue longer-term research aimed at resolving
key uncertainties. A motivation for the lower frequency of re-evaluation for OMPs is that substantive changes in the scientific understanding of developed fisheries only occur with the addition of several years of data, rather than just with the data for a single year [1]. This lower frequency may lead to concerns that the ‘reality check’ provided by conventional approach will be lost. However, the OMP approach does not preclude, but rather encourages, regular assessment updates and the use of alternative assessment assumptions (but not the full re-evaluations described above) as new data are collected.

The OMP approach recognises that a proper appraisal of risk for most fisheries cannot be made for a management decision that applies to a single year only, but needs to be based on the repeated application of a particular management policy over a number of years [1, 28]. It also allows for a lack of information to be taken into account, consistent with the principles of the precautionary approach [26]. In order to illustrate this, Cooke [28] plots utilisation (mean total catch) against information level ($1/[CV]^2$ of abundance estimates) for a given management procedure, which is tuned so that each point along the curve represents roughly the same ‘risk of depletion’ (Fig. 15.3). Figure 15.3 shows that when the available information is low, uncertainty over stock size is great so that TACs need to be set low. On the other hand, more information leads to better utilisation, although the marginal returns diminish as the quantity of information increases. Therefore, within the MPEF, there is a positive relationship between information and utilisation, and, in fact, an optimal level of information [56, 57]. In contrast to the implications of Figure 15.3, the conventional approach has tended not to limit harvest levels until there is sufficient information to indicate the need for such limits. Under these circumstances, information has a negative value to the fishery in the short term [28]. The precautionary approach (a changed burden of proof) was developed especially to overcome this problem.

![Fig. 15.3. Relationship between an index of utilisation (mean total catch) and information level ($1/[CV]^2$ of abundance estimates) for a management procedure, where each point along the curve represents roughly the same 'risk of depletion'. [Adapted from Cooke [28]; reproduced with permission of the ICES Journal of Marine Science.]](image-url)
The OMP approach provides a framework for dealing with conflicting or very different (but possibly equally plausible) interpretations of data as a result of model structure uncertainty. This is not possible under the conventional approach because of the necessity of selecting a 'best' assessment. For example, the ICES assessment of the northern hake stock (\textit{Merluccius merluccius}) produces somewhat different biomass trends depending on specifications for the tuning period, the plus group age, and F-shrinkage [58]. The ICES Working Group dealing with this stock is unable to resolve these issues within the current framework (where the conventional approach is used).

On the other hand, Geromont et al. [11] offer an example where industry argued that research survey estimates of abundance for Namibian hake were relative indices only and therefore that the resource was under-utilised and hence that TACs should be doubled. In contrast, government scientists argued that these estimates were absolute indices of abundance, leading to the conclusion that the resource was over-exploited and that TACs should be halved. The conventional approach again offered no solution to this problem. However, the OMP approach offered a way forward by identifying an OMP (based on trends in the abundance index) that allowed catches to increase over time if the industry view was correct, but reduced catches in response to downward trends in the index, thereby ensuring protection against further resource depletion, if the government scientists’ view was more appropriate [59]. This demonstrates a key feature of the OMP approach; the objective of an OMP is not to try to find ‘scientific truth’ (in this case whether the surveys are absolute or relative), but rather to use information to determine management actions in a way that is robust to scientific uncertainty.

An important by-product of the OMP approach is that it provides a basis for prioritising research [7, 59]. For example, an OMP may have to be relatively conservative to perform adequately for all hypotheses regarding the dynamics of the fishery. However, if subsequent research shows that some of these hypotheses are implausible, it may be possible to adopt a less conservative OMP, leading to larger average catches for the same perceived risk. Research priorities could therefore be set so as to address those uncertainties that could feasibly be resolved, and have the largest impacts on anticipated OMP performance [59]. In this manner, research is re-focused on those issues of greatest relevance to improved management.

15.7 PRACTICAL EXAMPLES

This section outlines the experience gained in developing and implementing management procedures in three cases: (a) Icelandic cod in Iceland, (b) the Revised Management Procedure and the Aboriginal Subsistence Whaling Management Procedure by the International Whaling Commission, (c) a mixed sardine-anchovy fishery in South Africa.

15.7.1 Icelandic cod

In the early 1990s, the Icelandic cod stock was severely depleted and below biological reference points such as the maximum sustainable yield level [60]. It was therefore decided to develop a stock recovery plan and a long-term management policy. Following
an evaluation of biological and economic objectives through simulation, the Icelandic government adopted a harvest control rule in 1995 [46, 61].

Although interactions among species (cod and its major prey, capelin and shrimp) were included in the operating model, there was no consideration of observation or estimation error, and it was assumed that historical catches were reported accurately. Uncertainty was mainly restricted to the form of the stock-recruitment relationship and the expected level of recruitment. The OMP was not evaluated using simulation and was based on a Virtual Population Analysis that used total catch and CPUE series from both fisheries-independent and -dependent sources. The TAC was set using the following type of harvest control rule:

\[
TAC_{t+1} = \max[0.25(Biomass_t + Biomass_{t+1})/2; 155,000 \text{ tonnes}] \quad (15.1)
\]

Catch was constant if stock size fell below a threshold (620,000 tonnes), otherwise it was 25% of the average fishable biomass. The minimum catch resulted from claims made in the Icelandic media by politicians and others, that catch could not realistically be reduced below a minimum level. A minimum catch of 155,000 tonnes was chosen based upon simulations that showed that this level would ensure recovery with high probability (>95%). However, this was potentially a high-risk strategy because it is unlikely to be robust to uncertainty about stock status and population dynamics (e.g., bias and precision of abundance estimates, errors in catch histories and non-stationary in productivity or carrying capacity) [24, 25, 48].

After implementation, it was found that the stock size of cod had been overestimated consistently and recruitment was subsequently lower than expected. Also, despite exploitation rates having been reduced, they were still above the long-term target of 25%. Large reductions in TAC in two consecutive fishing seasons (2000/2001 and 2001/2002) after implementation of the harvest control rule led to the managers to request a review of the rule. Following the review, it was concluded that while the large reductions in TAC would have been difficult to have foreseen, managers must be aware of not just the consequences of a decision were it to be perfectly implemented, but also the consequences of alternative, unintended, outcomes. This illustrates the problems (as discussed above) with assuming that, with no information to the contrary, the future will be similar to the past. It also illustrates the importance of evaluating uncertainty with respect to the ability to predict dynamic processes.

Subsequently, simulations that included multi-species effects (interactions between cod, capelin, shrimp, two seal species and three species of baleen whales) and uncertainty regarding estimates of stock size (cod only; 15%) were conducted (Hjorleifsson, Marine Institute, Reykjavik, pers. com.). However, these simulations were limited due to time constraints, ignored density-dependence in cod growth, and only explored limited options related to takes of whales and seals. This work resulted in a proposed revision to the harvest control rule that removed the minimum catch, reduced the target harvest rate, and stabilised catches by making next year’s TAC a weighted average of this year’s biomass and TAC, i.e.,

\[
TAC_{t+1} = (0.22 \text{ Biomass}_t + TAC_t)/2 \quad (15.2)
\]
This rule was, however, never implemented. Rather a new rule based on Eq. (15.1) without the minimum catch, but with a constraint on inter-annual changes in TAC of 30 000t was selected based on ‘back-of-the envelope’ calculations (Hjorleifsson, Marine Institute, Reykjavik, pers. com.).

15.7.2 International Whaling Commission

The International Whaling Commission (IWC) has developed OMPs for two different classes of whaling (commercial and aboriginal subsistence). These two classes differ in their management objectives, their political acceptability, as well as the nature of the whaling operations concerned. A single OMP (the Revised Management Procedure, RMP; [2]) has been developed for commercial whaling of baleen whales, while a more flexible scheme involving case-specific management procedures has been developed for aboriginal subsistence whaling (the Aboriginal Subsistence Whaling Management Procedure, AWMP). These OMPs have been tested extensively using the MPEF. Considerable effort has been expended to quantify the management objectives and assign priorities to them, and to clearly define realistic data and analysis requirements.

Development of OMPs for commercial whaling was initiated after the moratorium on commercial whaling was declared in 1982, in part due to the perceived failure of the harvest control rule being used at the time to provide management advice (the New Management Procedure—NMP). The objectives of the NMP were to recover stocks to optimum levels, to protect them at 54% of their pre-exploitation size, and to provide a formal (rather than ad hoc) structure for the development of scientific management advice. However, the NMP was inadequate because it required estimates of quantities (such as the maximum sustainable yield level) for which there were insufficient data and did not recognise (nor account for) uncertainty. The nature of the harvest control rule (which reduced catches from 90% of MSY when a stock was estimated to be at least 60% of its pre-exploitation size to zero at 54% of this size) also led to major fluctuations in scientific management advice regarding catch limits. The inadequacies of the NMP were further compounded by the changes in the objectives of many of the members of the IWC; i.e., over time, the political acceptability of commercial whaling, even if sustainable, dropped.

There were no formal OMPs for aboriginal subsistence whaling prior to the development of the AWMP for the Bering–Chukchi–Beaufort Seas stock of bowhead whales (adopted by the IWC in 2002; [4]) although some fairly ad hoc guidelines had been used by the Scientific Committee of the IWC to provide management advice prior to this.

The RMP and AWMP include detailed guidelines for survey methodology and design:

- detailed plans have to be submitted in advance to the Scientific Committee for review;
- the Scientific Committee has oversight of the surveys (and may assign observers);
- there are data analysis and availability guidelines; and
- the raw survey data have to be made available to the Scientific Committee in advance of the meeting at which an estimate of abundance is to be approved.

Formal ‘Implementation Reviews’ are also fundamental to the functioning of the RMP and AWMP, and occur every 5th year under normal circumstances. Implementation Reviews
include a review of the information required to apply the OMP (i.e., catches, abundance estimates) and of other information to check if the simulations on which the OMP was based are still plausible. These reviews are planned to take two years in case the OMP is rejected based on new information and additional work to identify a more suitable OMP is required. There is also a provision for unscheduled Implementation Reviews if new information causes concern. Possible reasons for ‘triggering’ an unscheduled Implementation Review identified for aboriginal subsistence whaling are [3]:

- major mortality events;
- major changes in habitat;
- a dramatically lower abundance estimate;
- harvesting/hunter information; and
- changed biological parameters.

15.7.2.1 RMP

The RMP is a generic package of rules that could be applied to any baleen whale that is harvested on its feeding grounds [7]. Therefore, in principle, it could be applied to blue whales in the Antarctic. However, this OMP was designed to have a high probability of setting zero catches for depleted stocks since the focus for simulation evaluation has been on minke and Brydes whales. As well as basing the RMP upon realistic data requirements (e.g., catches and estimates of abundance), it was rigorously tested and includes a feedback management mechanism to ensure robustness to uncertainty. The objectives of the RMP were explicitly stated and prioritised, i.e.

- it should have an acceptable risk level that a stock not be depleted (at a certain level of probability) below some chosen level (e.g., some fraction of its carrying capacity), so that the risk of extinction of the stock is not seriously increased by exploitation;
- it should be possible to take the highest continuing yield from the stock; and
- it should provide stable catch limits.

There were no explicit fleet or economic hypotheses or objectives. Any economic objectives were deemed to be captured adequately by the desire for high and stable catches. The RMP was developed by five groups of developers who constructed candidate OMPs that were then evaluated against the management objectives using the following types of performance statistics:

- risk-related statistics (e.g., lowest and final stock size; the probability of whaling being inadvertently allowed when the stock size is below the NMP Protection Level of 54% of unexploited stock size);
- catch statistics (e.g., total, short-term and long-term catches); and
- inter-annual variability in catches.

The range of uncertainties considered during the development of the RMP is perhaps the most extensive for any OMP. The operating models used when evaluating the RMP can be divided into two categories: (a) generic, and (b) case-specific. The generic simulations
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(conducted between 1987 and 1992) examined a range of hypotheses regarding population dynamics processes, although the operating model was not conditioned on the data for any specific case. The generic simulations were used to evaluate the impact of, for example:

- population dynamics and productivity;
- abundance estimates (bias, precision);
- errors in catch histories; and
- catastrophic events and changes over time in natural mortality, productivity and carrying capacity.

In contrast to the generic operating models, the case-specific operating models were developed to capture the key uncertainties pertinent to the cases for which the RMP may actually be used. These uncertainties relate primarily to uncertainty about stock structure and incidental catches, processes for which it is almost impossible to identify generic hypotheses that are sufficient to encompass the entire plausible range. To date, case-specific operating models have been developed for the Southern Hemisphere minke whales, the north Atlantic minke whales, and the western north Pacific minke and Brydes whales. The RMP cannot be applied to a particular species and region unless case-specific operating models have been developed for that species and region, and the RMP is found to adequately robust to the uncertainties concerned.

The trials used when evaluating candidate OMPs were divided into several categories, the main two of which were ‘base case trials’ and ‘robustness trials’. The base case trials comprised a limited and not very severe set of trials used to assist in initial development of OMPs and to eliminate at an early stage any OMPs that performed very poorly. The robustness trials were designed to judge performance in relation to a wide range of uncertainties. The robustness trials included a balance of trials that were difficult and those that were relatively easy. Difficult trials tended to be those that made extreme assumptions, which by definition were believed to be less likely. The aim of many of the difficult trials was to investigate the behaviour of an OMP. However, the best OMP was not necessarily the one that performed best in the extreme cases. Each robustness trial generally changed only one of the specifications of the base case trials, but robustness trials that varied several aspects of the base-case trials simultaneously were also constructed. This is because it was possible that two factors handled with ease by an OMP when they occurred singly may cause a failure in combination.

These trials led to the selection of a Catch Limit Algorithm (CLA) from the range of competing candidates. The CLA was then subjected to a much more extensive set of trials to examine the effects of combinations of uncertainties. The CLA [2, 28] incorporates a very simple population dynamics model (a discrete-time version of the Pella–Tomlinson model) that does not require prior estimates of biological parameters as there is no attempt to mimic reality in the underlying model used for management. The assessment model is fitted to the catch and abundance data using Bayesian methods in which information from new data is deliberately down-weighted to limit the speed at which the OMP responds to new data. The catch control law sets a catch limit of zero if the estimated population depletion is less than 54% of carrying capacity (i.e., the Protection Level), but otherwise the catch limit is based on estimated stock productivity, current stock status and the
uncertainty in stock status and productivity. Although it is not possible to manage in the absence of data, what to do in the absence of survey data was included in the RMP by introducing a ‘grace period’ of up to 8 years where catch limits would still be allowed in the absence of survey data, but after this period catch limits would be reduced to zero.

A set of trials was run to evaluate the performance of the CLA to rebuild stocks at initial depletion levels relative to the carrying capacity (K) of 0.2, 0.3 and 0.4, with different productivities (corresponding to the ratio of MSY to the stock size at which MSY is achieved of 0.01, 0.025, 0.04 and 0.07) [7]. The results showed that, although the probability of whaling occurring before a recovering population had reached 0.54K was high for low levels of productivity, the catches allowed would be so small as to only marginally affect the rate of recovery. Substantial reductions in the probability of whaling on stocks depleted below 0.54K could be achieved, but this would lead to only slight improvements in recovery rates and would be at the expense of substantially lower catches.

The RMP as originally developed as a single-stock OMP, but whaling may occur on a mixture of whale populations. For example, coastal type whaling might impact a range of stocks as they migrate through an area. Therefore, the RMP contains case-specific rules to handle uncertainty regarding stock structure. Specifically, catch limits are set for geographic areas referred to as ‘Small Areas’. A Small Area is defined as being small enough to contain whales from only one biological stock, or if whales from more than one biological stock were present in the area, catching operations would be unable to harvest them in proportions different to their relative abundance in the area. Although this approach should be adequate to handle stock structure uncertainty, it has proven difficult to identify an appropriate range of stock structure hypotheses given difficulties in interpreting data, particularly genetic data. The main problems in trying to implement the RMP in practice at present therefore relate to identifying alternative stock structure hypotheses, determining which such hypotheses are plausible, and ranking the plausible stock structure hypotheses.

15.7.2.2 AWMP

The AWMP differs from the RMP because it is intended to deal with hunts on a case-specific (rather than generic) basis. The development of the AWMP started with two ‘data-rich’ cases (the Bering-Chukchi-Beaufort Seas stock of bowhead whales and the eastern North Pacific stock of gray whales). In recent years, focus has shifted to two more ‘data-poor’ (and hence difficult) cases, fin and minke whales off west Greenland.

Unlike the RMP, the AWMP is not used to set quotas directly, but rather to ensure that quotas, set in response to a statement of need by the aboriginal whalers, meet management objectives, which are:

- ensure risks of extinction not seriously increased (highest priority);
- enable harvests in perpetuity appropriate to cultural and nutritional requirements; and
- maintain stocks at highest net recruitment level, and if below that, ensure they move towards it.

This means that the AWMP does not aim to maximise catches or economic objectives. Rather, aboriginal whalers determine the number of whales needed to satisfy their cultural
and nutritional need. A statement of need (in number of strikes, and not landed whales) is submitted to the IWC. This need becomes the catch limit if the need statement is accepted by the Commission and the strike limit calculated using the AWMP for the species concerned equals or exceeds the need. The AWMP produces a strike limit rather than a catch limit (using a ‘Strike Limit Algorithm’, SLA) because not all whales struck in aboriginal hunts are successfully landed. However, the AWMP assumes that all strikes result in kills. Because the IWC recognised that the Arctic environment could result in difficult hunting and survey conditions they requested that provision be made for block quotas (i.e., quotas are set for 5 years) and to allow for the carry-over of quotas. Furthermore, the ‘grace period’ for aboriginal subsistence whaling exceeds that for commercial whaling.

During the development of the AWMP, scientific discussions focused on what performance measures to use, the specification of the simulation trials, and the selection process. The development of the AWMP was also based on several categories of trials, and evaluated the same types of factors as were evaluated for the RMP (although these were tailored to the specifics of the species under consideration). In addition, different survey frequencies, different historical and future survey biases, and differences between true and estimated CVs were also evaluated. Candidate SLAs were again developed through a combination of competition and cooperation, using experience gained from development of the RMP. For example, in relation to the bowhead stock, five ‘candidates’ were developed by 2001, resulting in two being presented to the IWC in 2003 and the selection of a single candidate in 2004. The Bowhead SLA adopted by the Commission involved averaging the results from two different SLAs as this improved robustness.

The operating models on which the OMP evaluations for the bowhead and gray whales stocks were based were age- and sex-structured and were conditioned by fitting them to catch, abundance and, in the case of the bowhead stock, the proportions of the population that consist of calves and mature whales [3, 6]. The fitting procedure was Bayesian, with the priors for the parameters of the operating model determined by the Scientific Committee of the IWC. Although the operating model was conditioned on the existing data, a variety of assumptions regarding the future that are not directly supported by existing data (e.g., the occurrence of catastrophes) were also investigated, although care was taken not to select an SLA because of its superior performance for a scenario that was not considered particularly plausible.

A 100-year time period was used for the simulations. However, future need depends on growth of the human population rather than that of the stock (cf. RMP) and so there was a requirement to perform evaluations using a ‘need envelope’ to set bounds on future need. For example, the IWC agreed that the upper bound of the need envelope for the eastern North Pacific gray whales would be 530 strikes per annum [6]. New trials have to be conducted if a future need request exceeds this maximum.

15.7.3 South African sardine and anchovy

South Africa developed an OMP for the mixed sardine-anchovy fishery [10], for which one of the main drivers is the bycatch of juvenile sardine in the anchovy directed fishery. This bycatch has a major impact on the higher value directed sardine fishery. Three fleets are considered, one targeting adult sardine, one targeting anchovy (with an associated
bycatch of juvenile sardine) and one targeting round herring (with an associated bycatch of adult sardine). Only the directed anchovy and sardine fisheries are modelled explicitly, while for the round herring fishery it is assumed that a fixed tonnage of sardine is taken every year, so that the population dynamics of round herring is not included in the operating model.

A range of alternative population dynamic hypotheses were considered, including different natural mortality values for juveniles and adults, alternative forms for the stock-recruitment relationship, increased recruitment variability to reflect the possibility of future environmental conditions causing recruitment to become more variable, and a halving of carrying capacity over time to reflect the possibility of ‘negative’ regime shifts. Alternative hypotheses were also considered regarding the data sampled from the operating model, which included different ageing methods for anchovy, use or not of survey proportion-at-age estimates for sardine, and alternative assumptions regarding survey bias and additional variance (including density dependence) for both stocks. No economic considerations were included in the development of the OMP; only proxies for economic quantities were used and included as summary statistics.

The operating model for the South African sardine and anchovy stocks was conditioned on the basis of maximum likelihood fits to survey indices of abundance of recruitment (age 0) and spawning biomass (with associated sampling variances) and spawning biomass survey proportions-at-age, with catch-at-age data assumed known without error.

Process error was included only for recruitment, which was generated as lognormally distributed, auto-correlated deviates from a stock-recruitment relationship. Observation error in the data sampled from the operating model was included as lognormal error for survey indices of abundance for recruitment and spawning biomass (with appropriate bias), where allowance was made for both sampling and additional variance components. Observation error was also included as lognormal error for estimates of sardine to anchovy commercial catch ratios. In both these cases, correlative relationships in residuals were preserved, between sardine and anchovy survey estimates for the former, and between successive months of sardine bycatch to anchovy catch ratios for the latter.

Estimation error for the parameters of the operating model took the form of a Monte Carlo simulation of the joint probability density function for these parameters, obtained using parametric bootstrapping. Model error was explored in terms of the alternative hypotheses about population dynamics processes/variables on the one hand, such as natural mortality, carrying capacity, recruitment variability, and the stock–recruitment relationship, and data generation processes on the other, such as the form of the additional variance component associated with the survey indices of abundance. Implementation error allowed for the difference between the total allowable bycatch set for sardine, and the actual bycatch taken in the anchovy-directed fishery, where the actual bycatch depends on the sardine to anchovy commercial catch ratios in successive months in the fishery. The anchovy fishery is closed when the actual (i.e., operating model) sardine bycatch exceeds the total allowable sardine bycatch, so that under these circumstances the anchovy catch will be less than the corresponding TAC.

Monitoring of catches is assumed to reflect true catches perfectly, because, although discarding does occur, it is assumed to be negligible. Enforcement of TACs is assumed to be effective in that TACs are always assumed to be taken, except in the case of the
Operational management procedures

bycatch allowance set for sardine, which can both be exceeded or not taken, but when exceed can lead to the anchovy TAC not being taken.

For the South African sardine-anchovy OMP, stock status is derived directly from spawning biomass and recruitment surveys, and therefore does not rely on an assessment model (i.e. it is a model-free approach). Harvest control rules for the South African sardine-anchovy fishery encompass management of both stocks within the same framework because of the technical interactions between them. These rules rely on survey estimates of abundance for spawning biomass and recruitment and survey mean weight-at-age, as well as in-season estimates of commercial catch- and mean weight-at-age, and sardine to anchovy commercial catch ratios. The rules also include constraints on year-to-year and within-season adjustments to TACs, and further refinements such as incorporating rights-holder preferences for sardine to anchovy catch ratios [62]. These rules have evolved over time to a high level of complexity, which has become necessary to maintain the flexibility needed for optimal utilisation of the sardine and anchovy stocks.

15.8 DISCUSSION

OMPs, as defined above, were first developed by the International Whaling Commission following the declaration of the moratorium on commercial whaling in 1982. The previous management system (based on the NMP) is perceived to have failed despite the fact that its objectives appear to be consistent with good management practice, i.e., it was designed to bring stocks to an optimum level, and to provide a formal advice structure. Indeed, the NMP objectives are broadly comparable with the advice framework of the Advisory Committee on Fisheries Management (ACFM) of ICES, which is to ensure that spawning stock biomass (SSB) remains above a threshold \( B_{lim} \) at which recruitment may be impaired and that fishing mortality remains below a threshold level \( F_{lim} \) that would drive the stock below the biomass threshold.

The perceived failure of the NMP was because it required estimates of parameters that could not be obtained using the available data and because it ignored uncertainty. The lessons learnt from the NMP were that

- objectives must be stated explicitly and assigned priorities—often these are stated too broadly and are difficult to convert into quantitative measures;
- data and analysis requirements must be realistic and specified;
- limitations of the advice framework and regulatory regime must be recognised and the inevitable uncertainty explicitly taken into account, including considering what we do not know as well as what we do;
- management tools should be rigorously tested (e.g., using computer simulations); and
- there is a need to be explicit about how to detect and respond appropriately to changes in stock size and/or population dynamics.

There are many lessons for fisheries management in general, and the current ICES advice framework in particular. The ICES framework is primarily focussed on a single-species basis, assumes that reference points represent stationary processes, and neglects to test
the robustness of management advice against the full range of uncertainty. Kell et al. [25] showed for North Sea cod that error and bias in stock assessment methods, when using harvest control rules or conducting recovery plans, is complex and difficult to second-guess. The Icelandic example emphasises the point that harvest control rules should be fully evaluated as part of a OMP within a MPEF incorporating uncertainty related to process, observation, estimation, model and implementation error. Such evaluation should be used to identify OMPs, including appropriate biomass trigger points and catch levels, that are robust to the definition of stocks, possible changes in productivity of stocks over time, and non-compliance with regulations (i.e., implementation error), as well as the ability to monitor and control stocks.

There appears to be a lack of separation between the role of managers and scientists in the case of the ICES scientific advice framework, where scientists have often taken upon themselves the role of defining precautionary reference points. While scientists have a role in defining limit reference points (e.g., based upon biological models) the definition of precautionary reference points depend on what is considered to be an acceptable level of risk and this is not a scientific, but rather a socio-economic, issue. For example, many species are taken in mixed fisheries, where current single-species management objectives are often conflicting and encourage discarding, high-grading, and/or black landing.

There could also be an incentive for the industry to improve its performance if precautionary biological reference points take account of implementation and estimation error. Specifically, the OMP evaluation approach could be used as a planning tool in fisheries negotiations to demonstrate that improved performance may lead to higher catches in the short term. This is because there is a positive relationship between information and yield in this approach (Fig. 15.3) so there is an incentive to resolve key uncertainties about the population dynamics to avoid having to reduce exploitation rates to be robust to those uncertainties.

The process of conditioning operating models on available data and knowledge ensures that the OMP approach is firmly rooted in the existing knowledge base, which is important in actually being able to agree on an OMP. However, there is also a need for a review process to handle cases in which either new knowledge becomes available, the stock or fishery parameters fall outside of the range originally evaluated, or the OMP behaves in a way not previously envisaged. There is a tendency to limit the hypotheses included in OMP evaluations to those with considerable support. However, the OMPs suggested using such operating models may not be robust against unknown, and perhaps unquantifiable, risks. Unfortunately, it is not uncommon for factors that have never impacted a specific fishery, but have impacted other fisheries, to suddenly impact that fishery. Ignoring information on what might occur based on data for other fisheries should therefore be avoided. At the same time, care should be taken not to base the selection of an OMP solely on an extreme, but highly unlikely, scenario.

Although the OMP approach is a powerful tool, ultimately the aim is to improve the quality of management. Importantly, the OMP approach is intended to improve the current system, not by making it more complex, but by identifying a robust management framework that can handle the often conflicting and poorly defined management objectives, account for many of the uncertainties that are often ignored in the conventional approach, and aid in strategic decision making. Developing and implementing an OMP will not
necessarily improve the quality of assessments, which may still need to be conducted on an annual basis. Rather, it will provide a way by which management can take the unavoidable uncertainties and possible disasters of assessment methodology into account. In this respect, it links together the management and assessment frameworks, and clarifies the roles of the main players, as is demanded in several jurisdictions (e.g., the CFP of the European Union).

An additional benefit of the OMP approach is that it can be used to perform cost-benefit analyses and ensure that resources are allocated appropriately to meet management objectives. The cost of computer simulation is much less than the cost of collecting data and the value of forgone yield due to bad management. The approach has successfully been used for small stocks, for example, the Blackwater herring [39], which enabled assessment and management costs to be reduced and for the stock to achieve Marine Stewardship Council certification.

The OMP approach requires the development of special software to implement the operating model and the model for data generation. Until recently there were no software packages that implemented ‘generalised’ operating models. However, a generic framework FLR [63] has now been developed by the FEMS EU project. Use of this package should increase the ease with which the OMP approach can be applied to new fisheries.

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REFERENCES

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Chapter 16

Management Strategy Evaluation (MSE) and Management Procedure (MP) Implementations in Practice: A Review of Constraints, Roles and Solutions

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16.1 INTRODUCTION

During the last two decades, the role of fisheries management has substantially changed. Poor outcomes from traditional management systems have encouraged scientists and managers to design new tools to manage fisheries. The aim of developing these tools is to abandon the old schemes of prescriptive management and move towards a descriptive management approach. As a consequence, there is an increasing trend among scientists and fishery managers to develop a management approach that enables evaluation of the performance of alternative management strategies for pursuing different management objectives. These systems offer a basis for managerial decision-making by running simulations that offer a snapshot of the trade-offs between the various management objectives and test the robustness of the system to uncertainty. Thus, the use of these systems is consistent with the precautionary approach to fisheries management [1].

Currently, there are a few of these systems in operation around the world. They are quite similar and hold much in common with regard to their philosophies and methods. In the technical literature these systems are categorised as management procedures (MP) [2]; management strategy evaluation (MSE) [3] and harvest strategy evaluation (HSE). According to Sainsbury et al. [4], these systems feature the adaptive management approach developed by Walters and Hilborn [5] and Walters [6] as a common antecedent. The International Whaling Commission (IWC) was the first institution to pioneer the development of MPs.

The MSE approach is slightly wider in scope than the MP and runs evaluations that do not necessarily deal with feedback harvest strategies. Also, its purpose is not necessarily to develop an agreed management procedure, but to provide an objective basis for short or long-term decision-making [3]. The most relevant examples are the MSE carried out in Australia [3] and the MP implemented in South Africa [2]. In the technical literature, the MP carried out in South Africa is categorised as an operational management procedure...
(OMP) (see the introduction of this section). These systems are under the glance in Europe. Various studies that evaluate management strategies have been carried out on behalf of the European Commission. For example, Kell et al. [7–9] focus on evaluating restrictions in inter-annual variations for TACs.

These innovative approaches have as an essential component the joint participation of managers, scientists, industry representatives and other stakeholders in data collection, assessment, advice and decision-making. Since the success of these approaches relies heavily on participation, the enhancement of communication between managers, scientists, industry and other interested groups is a key task to deal with. During the implementation process, difficulties arise because parties without a scientific/technical background find the system as a whole and the underlying models cryptic. Moreover, the concept and the definition of uncertainty are usually misunderstood. The role of scientists in strengthening communication through the use and development of diverse means is essential. Effective communication and cooperation is a cornerstone that will help to develop hypotheses about the functioning of the fishery system and its relationship with the ecosystem. Good communication effectively brings user knowledge into the process and encourages their active participation in the implementation process.

The implementation of these systems is usually resource demanding—it requires changes in the management apparatus, improvements in technical capabilities and skills, and large budgets—and they have to face considerable constraints presented by, for example, the unwillingness of managers and other relevant people to change management scenarios, and the communication gap between scientists and fishers. In spite of these constraints, these approaches to management seem promising and major researchers encourage moving towards them [10]. Among the positive outcomes expected from such an implementation are increases in the transparency of management processes, the incorporation of stakeholder participation and knowledge, each of which strengthen legitimacy and, in turn, incentivise compliance. In this section, we will not attempt to describe the technical aspects of these approaches but to offer an overview of the constraints and problems encountered in the implementation of some of the systems currently in use, focusing on systems that have been in use and widely documented during the last fifteen years, particularly on the South African OMP, the Australian MSE and, to a lesser extent, on the New Zealand rock lobster MP and the American Pacific salmon HSE. Issues key to this discussion are industry participation and knowledge inclusion and the role of scientists as conductors of the process.

16.2 MSE AND MP IMPLEMENTATIONS: THE CHALLENGES

16.2.1 Conflicting objectives

Multiple and frequently conflicting objectives are commonly encountered in fisheries management [2]. These differences usually generate conflict between groups, who prioritise different management goals—for example, the classic conflict between those interested in resource protection and those seeking to increase the exploitation rate because of socio-economic interests. Conflict is also observed between user groups aiming at the same
objective. For instance, users aiming to maximise economic returns, but using different means to achieve it—examples include artisanal fishermen versus industrial fishers and fishers versus processors.

Conflict complicates the implementation process of a MSE/MP, since clashing positions are not only found between scientists and industry but are also encountered between factional groups within the industry. The case of the South African pelagic fishery is a good example of the implementation of a MP within a scenario featuring conflicting objectives. This is a multi-use multi-species fishery with two major species and two major user groups. During the implementation of the operational management procedure (OMP) in the South African pelagic fishery, conflict arose between industry, scientists and managers. While industry aimed to maximise catch and benefits, scientists and managers aimed to minimise the risk to the stock. Conflict also arose within the industry due to conflicting objectives relating to the needs of the anchovy-based reduction and sardine-based canning industries [2]. Barnes [11] suggests that conflict arose because not all members have a stake in both fisheries. Thus, there was little incentive to support a complicated two-species OMP that sacrifices a fisherman’s potential catches of anchovy to save another’s potential catches of sardine. Conflicting objectives may polarise groups in such a way that scientists are acknowledged as being the defenders of stock while industry is acknowledged as being exploiters of the resource, aiming for socio-economic objectives. During the implementation of the South African OMP, scientists had to face the accusation of being responsible for causing socio-economic distress and were appointed with the responsibility of having the power to alleviate it [2].

Although fishers are usually regarded as aiming to further increase high catches, Bentley et al. [12] report that, in a workshop held in New Zealand, lobster fishers highlighted the importance of goals such as stability in catch quotas, maintenance of high CPUEs and maintenance of a wide range of lobster sizes, so that fishers could respond to changes in market demand. As a result, they expressed a willingness to trade some potential catch for stability and abundance goals. Francis and Shotton [13] highlight the need to increase dialogue between the different parties to clarify objectives. However, the definition of objectives can be a complex task, since the increasing incorporation of industry and other stakeholders—for example, conservationists, indigenous people and recreational fishers—in decision-making and other processes of management turn management into a highly complex task.

16.2.2 Resistance among decision-makers

Within a fisheries management organisation, the implementation of innovative approaches to management such as MSE/MP is expected to face scepticism and severe constraints. This is because, first, it may require changes in the management apparatus, such as changes in the hierarchy, acquisition of expertise and improvements in technical capabilities and infrastructure. Therefore, time and funds are required to both reshape and equip organisations. International experience shows how the reluctance to change of people involved in management acts as a powerful constraint to the introduction of these approaches. Smith et al. [3] report that, in Australia, managers were unwilling to accept a system that requires explicit and measurable performance indicators. According to them, these
indicators were understood as a basis for auditable management performance. In addition, the set up of explicit objectives and procedures may be perceived by managers as a loss of the flexibility required to respond to emerging problems and to trade-offs across conflicting interests and objectives. In this context, it is not surprising that the perspective of the application of a MSE/MP may meet resistance, not only among managers but also among management officers. Managers may perceive it as a threat to their autonomy, while officers may see it as an obligation to learn, effectively and in a relative short time, unfamiliar and rather complicated procedures and techniques.

16.2.3 Lack of technical and economic resources

A lack of technical skills in stock assessment, and modelling in general, acts as a powerful constraint to management, even in developing countries. Hilborn [10] reports a lack of modelling skills in USA, which is perceived as a major impediment to the provision of scientific advice. In the case of MSE/MP, this problem is magnified because the operating models are often markedly more complex than the models underlying most stock assessments. Moreover, these approaches require the development of special software. As Butterworth and Punt [14] report, no general software packages are available to implement ‘generalised’ operating models. Thus, the difficulty in developing such specialised software is also an impediment to a wide application of a MSE/MP approach.

Budgetary factors are also important limitations to the implementation of MSE/MP. The implementation of these approaches require, inter alia, development of software, use of powerful computers, training of staff, hiring of consultants, improvement of the data collection system, and organisation of informative sessions. These requirements may test severe budget constraints. Considering that lack of funds is a powerful constraint, it is necessary to carefully evaluate the cost-effectiveness of such an introduction. The cost-effectiveness of these systems is somehow a matter of scale: for example, it is not worth introducing such a resource-demanding approach into a small-scale fishery [1]. In cases of limited budget and lack of technical capabilities, Butterworth and Punt [14] suggest carrying out the framework without complex robustness testing processes. Taking into account the aforementioned constraints, the management procedure approach seems unlikely to replace conventional management in a global context. Caddy and Mahon [15] point out that the costs and availability of information, capabilities and expertise to carry out management in the face of uncertainty may impede the use of these systems in small-scale fisheries and for most fisheries in developing countries. Conversely, the OMP approach may be advisable to reduce the costs of managing small scale fisheries in developed countries. Roel et al. [16] evaluates an OMP for the Black water herring of the Thames Estuary. This fishery has experienced a reduction of economic value due to changes in consumers’ preferences. Traditionally, this fishery has been managed by an annual TAC based on the results from an Extended Survivors Analysis (XSA). Since the annual process is highly resource-demanding, the managers have requested the evaluation of multi-annual TACs that may reduce the costs of undertaking annual studies.

To sum up, these innovative approaches are expected to play a key role in managing fisheries in the near future [10]. Although costs may be too high for governments to bear, it is possible for the burden of implementation to be shared with, or even entirely borne
by, industry. In Australia, for instance, the costs of the MSE approach are entirely met by the industry. This enhances the involvement of the industry in the management process.

16.2.4 Stakeholder reluctance

The MSE/MP approach brings active stakeholder participation into the process of fisheries management, which has traditionally been an area closed to resource users. Despite the fact that this participatory characteristic may be palatable to most stakeholders, such an implementation may face some opposition from stakeholder groups. Such opposition can be explained by resistance from stakeholders to the requirement to assimilate new concepts and to accept an approach that is neither very well known, nor has yet widely demonstrated its effectiveness [3]. It can also be unpalatable for stakeholders to devote time and effort to deal with issues that do not address topics such as allocation and others that may interest fishers more. Other factors that may make stakeholders unwilling to participate is the open and public nature of the process that somehow obliges them to assume a unified position, regardless of the fact that factional groups may have conflicting interests and clashing positions, and to compromise on harvest rules, which are set well in advance of fishing activities. The underlying rationale to such reluctance is that this type of management system may reduce the flexibility that makes traditional management prone to pressure (for example, by undermining scientifically recommended TAC reductions). This reluctance is quite natural since stakeholders will be forced to abandon the management system within which they have been working for decades. However, experiences in Australia and South Africa have, in most cases, shown that stakeholders have positively responded to the challenges and sympathetically joined the change.

It seems that industry acceptability is partly determined by the scope of planning horizons. This fact is especially noticeable in South Africa where the pelagic industry, dominated by a few large companies with a long-term planning horizon, widely accepted the system. Conversely, the same system would not have been so easily accepted by an industry composed of a large number of small operators that have cash flow considerations and, thus, short-term planning horizons [17]. In countries with an ongoing property rights system, industry acceptability of MSE/MP may depend on the quality of the property right fishers hold. Long-lasting fishing rights and security of ownership help industry to set a long-term planning horizon. Duration and security of rights may partly determine their willingness to shift to a system like a MSE/MP and their degree of involvement in the process. The rationale is that industry may not fully accept the implementation of these systems in a fishery in which changes in policy may determine changes in the ownership of rights. They may consider this situation before making an investment in terms of both effort and funds. Butterworth and Punt [14] report that, in South Africa, at the time of the big post-apartheid political change, there was particular concern regarding the allocation of rights to formerly disadvantaged groups. Even though the industry has supported the OMP implementation, within the process uncertainty about the changes in right tenure played some role in the choice of shorter term procedures with highest immediate catch levels.
16.2.5 The communication gap

Participation being an essential element of MSE, the achievement of an understanding between scientists, managers and stakeholders is a key factor of success. Cochrane [18] sees that, when managing by strategies, communication is essential at all levels, giving all the parties responsibility for ensuring that accurate information is provided and correctly interpreted. The problem is that an understanding between groups who have different educational backgrounds and objectives may be difficult to reach. These differences generate poor communication, misunderstanding and mistrust among the parties, which undermine the performance of the approach. In fact, communication problems between scientists, managers and industry are important sources of uncertainty [19]. Since scientists and non-scientists do not speak the same language, arriving at a consensus is difficult. In this circumstance, some creative means have been developed to speed up communication between these two groups. Developing the means to ease communication between diverse groups is a task for cognitive psychology and social and management sciences [19].

Even though the underlying philosophy and the overall functioning of these systems are not hard to comprehend, particular aspects of the approach have cryptic results and/or are hard to assimilate for people without a scientific/technical background. For example, Bayesian statistics are particularly difficult to follow and comprehend [3]. In the case of the South African pelagic fishery, one of the initial problems faced during the implementation of the OMP was the difficulty in explaining the stochastic nature of the models used to the industry groups as a whole, as well as clarifying and defining how ‘risk’ is measured and the level of acceptable ‘risk’ set. This was a problem encountered even among members of the industry that held an engineering background [2]. It is worth highlighting that complexity of models and evaluation frameworks is not only a problem for non-specialists. Hilborn [10] let us know about a lack of these skills in such a developed country as the USA. According to him, growing complexity and internal assumptions make it difficult to understand what drives many assessments. Moreover, when assessment becomes more complex the models are no longer open to the simplifications that enable understanding. Thus, the increasing complexity of models reduces their accessibility and fewer people are able to understand and apply them effectively. De Oliveira and Butterworth [20] report that the problem of ‘accessibility’ is particularly noticeable in the case of the South African pelagic OMP. In this case, modelling procedure has evolved into a complex and cryptic system that can be compared to a ‘black box’, which is seldom understood by managers and industry in spite of their active participation in assessment.

16.2.6 The difficulty of including users’ knowledge in scientific assessment

Although user participation has been widely welcomed within the scientific community, users’ knowledge inclusion in a MSE/OMP is a challenge. Baelde [21] reports that one of the constraints encountered during the process of implementing MSE in Australia was the technical difficulty of integrating fishers’ knowledge into the process. According to this author, the main obstacle faced by scientists wanting to integrate fishers’ knowledge is how to quantify it. Fishers’ knowledge is acknowledged as being mostly qualitative and narrative or anecdotal, holistic rather than sectoral and subjective rather than objective.
Fishers’ knowledge reflects not only the environmental, biological and socio-economic scenario in which fisheries operate, but also personal beliefs and values. Faced with the inability to quantify fishers’ knowledge, scientists tend to question the usefulness of that knowledge. On the other hand, fishers experience frustration at scientists’ inability to make use of industry information and views. Smith et al. [3] consider that scientists need to be open to the consideration of fishers’ knowledge in the form of non-standard hypotheses and non-standard data and need to find ways of incorporating these into the formal process of quantitative assessment and evaluations.

Besides methodological difficulties, with regard to quantifying and including fishers’ knowledge into the assessment, which are encountered, there are other factors that may determine the reluctance of scientists to take fishers’ knowledge into account. These are socio-cultural barriers that hamper communication between scientists and fishers. These barriers are founded in the perception that science holds a moral authority while fishers’ knowledge is suspicious of being biased by vested interests. These barriers provoke scepticism among scientists regarding the validity of fishers’ knowledge. Overcoming the above mentioned technical and socio-cultural barriers is one of the main challenges in implementing MSE in the context of open and participatory assessment and evaluation groups. This challenge requires active multidisciplinary research [21].

16.3 THE ROLE OF THE DIFFERENT PARTIES DURING THE IMPLEMENTATION PROCESS

16.3.1 The role of the institutional setup

Active inclusion of stakeholders in the management process is a key factor of success in fisheries management and provides a fertile field of development for MSE/MP approaches. In this context, the Australian case is a good example of participatory management. The co-management approach implemented by the Australian Fisheries Management Authority (AFMA) stresses stakeholder participation in all key areas of fisheries management, such as setting objectives, data collection, research priorities, stock assessment, enforcement and decision-making. The final decision within the AFMA management system is taken by the AFMA Board. This board includes representatives from resource management organisations, industry, research and conservation groups. For each fishery, the AFMA Board is supported by the Management Advisory Committees (MACs), the membership of which also includes representatives of the AFMA management, industry, research, recreational fisheries and conservation groups.

Within the MACs, issues of concern are debated, problems identified, possible solutions discussed and developed and recommendations forwarded to AFMA. MACs act as a forum in which industry exposes its views to AFMA, AFMA consults industry on management arrangements, and consultation is facilitated between industry and scientists, as well as enabling information to flow between the parties involved. The AFMA board is also assisted and advised by Fisheries Assessment Groups (FAGs). These groups consist of AFMA members, scientists, economists, industry members, conservationists and
recreational fishing representatives. FAQs are independent of MACs and responsible for undertaking stock assessment in each commonwealth’s fishery [3, 22].

16.3.2 The role of scientists in the process of implementation

Science is commonly acknowledged as being an objective and essential support for political decisions concerning natural resource exploitation. Even though public opinion holds that science should be independent of political imperatives, scientific knowledge is, in most cases, produced under governmental mandate within a rigid ‘top-down’ management framework. Therefore, it is hard to guarantee that scientific advice provided to government will be followed by it. In a MSE/MP-like approach, the participatory setting of objectives, the open process of stock assessment and the transparent setting of measures somehow guarantees ‘no-pollution of science’. That science has a high prestige among most of the public and managers, and considering that the MSE/MP approach was born within the scientific community, it is advisable that scientists should steer the implementation process. They have the greatest knowledge of the potential yield, trends and variability in stock levels and of the more cryptic aspects of the framework, and, thus, it is necessary that they instruct managers and stakeholders in all the aspects of the process.

Traditionally, formal knowledge production has been conducted by natural scientists. Currently, there is a tendency to incorporate socio-economic knowledge in the process of management. According to Smith et al. [3], the MSE approach requires the use of all available sources of knowledge and scientific disciplines. The role of economists and sociologists is meaningful throughout the process—for example, in the modelling of fleet dynamics—but especially in estimating the cost effectiveness of the approach and finding the means to enhance legitimacy and, therefore, compliance.

Scientists’ participation is also fundamental when presenting the system and their results to the rest of the participants. Enhancing communication by presenting information in a transparent and understandable way is one of the challenges to the scientist. In South Africa, efforts by scientists to ‘educate’ industry and managers in the functioning of the OMP approach have been widely welcomed. In the particular case of the South African rock lobster fishery, industry acknowledged the efforts made by scientists in instructing industry during the implementation process and called this period ‘the scientific renaissance’ of the fishing industry’ [23].

16.3.3 The role of stakeholders: Stakeholders’ knowledge inclusion and active participation in the process

Several authors from a variety of disciplines point to the importance of bringing stakeholders knowledge into the process of management ([18, 24–27]) User participation brings a stream of expertise and knowledge into the management process, which, in turn, generates legitimacy and compliance. Moreover, it generates a diversified and useful understanding of the environment, the interrelations between resources and men and a comprehensive understanding of fishers’ behaviour and technological factors. This knowledge is updated and enriched in the fishers’ day-to-day professional activities. Modern management relies on stakeholder participation to step away from the traditional command and control
User participation is essential when carrying out MSE/OMP approaches. Butterworth et al. [17] point out that the underlying philosophy of an OMP demands that managers, scientists and industry agree on clearly defined rules prior to its implementation. This participatory setting of rules is one of the main components of these systems. It encourages participants to define objectives and plans and present and discuss them with the other participants. Smith et al. [3] report that, in Australia, the co-management approach is based on the premise that the most successfully managed fisheries, in terms of both sustainability and economic returns, are those that include the skills, knowledge and expertise of stakeholders in the fishery management process. On the other hand, C&C regimes have produced, inter alia, a lack of understanding of both commercial realities and the day-to-day environment. As a consequence, enormous distrust and even hostility between fishers, scientists and fisheries managers impede effective management. Since the Australian MSE approach requires active participation, the very well established Australian co-management system has offered the MSE approach a fertile field of development.

Despite the fact that these approaches to management are considered as state-of-the-art systems and acknowledged for their use of sophisticated analytical tools, usually cryptic to non-scientists, they demand user participation at a variety of stages in the management process. In this context, it is remarkable how some countries have managed to bring active user participation into the system, demonstrating that this is a perfectly achievable goal. Involvement of stakeholders begins with collaboration on data collection—something that can be called the primary level of participation. Moreover, their perspectives regarding the interaction between environment, species and men are considered of capital importance. Stakeholders play an important role in helping to develop the simulation-based operating models, which attempt to mimic reality. These virtual scenarios are built from hypotheses developed using available data and expert opinions from scientists, managers and participating groups [21]. Cochrane et al. [2] report that, in the MP carried out in the pelagic fishery of South Africa, industry participation allowed scientists to get in touch with the problems of the industry and to be aware of their needs. Stakeholder inclusion in the process produces transparency that brings about legitimacy. In this regard, the International Council for the Exploration of the Sea (ICES) [28] Study Group on Management Strategies point out that management strategies cannot be implemented in an open society if they are not considered legitimate by stakeholders and citizens. The active participation of stakeholders in the process of implementation substantially increases transparency. Moreover, opening the processes to public scrutiny (for example, through hearings) increases transparency.

The management of salmon in Washington (USA) is a good example of users’ participation in an innovative approach to management. The management of this fishery is acknowledged as one of the most complex resource management challenges in USA, due to the complex interplay between biological and geographical factors. Management is backed up by a system of data collection in which users (in this case aboriginal communities) bring data, which is fed into a computer model that simulates a snapshot of an upcoming season’s fishery under various regulation options. The results from these
simulations are, in turn, compared to conservation objectives, obligations under USA-Canada treaties, allocations for tribes, and protection requirements for some wild fish populations under the Endangered Species Act. In this fishery, stakeholder participation is a key factor for the setting of conservation objectives. These objectives are jointly set by state and tribal fish managers. The season setting process occurs in a series of public hearings in which concerned federal, state, tribal and commercial fishing industry actors participate [29].

Another good example is the case of the New Zealand rock lobster MP, which is analysed Bentley et al. [12, 30]. They report the inclusion of stakeholders as diverse as commercial, recreational, customary and conservationist in the process of agreeing management objectives and developing associated performance indicators for the fisheries. The analysis of the trade-offs between different management objectives was one of the most important issues during the design of the system, with the different weight assigned to each objective having a determinant role in defining the procedure.

In Australia, the incorporation of other stakeholders as conservationist agencies into the partnership approach has faced some constraints. For instance, conservationists are unfamiliar with the assessment methods. Moreover, conservationists’ belief that fisheries management has almost universally failed has provoked scepticism directed at management and the assessment that supports it. Apart from these constraints, the incorporation of conservationist agencies into the partnership approach has been positive, providing useful input into the implementation of the MSE and helping to maintain wider public support for the assessment and management process. It has also helped to take some of the heat off the scientists that otherwise tend to be seen by the industry as ‘green’.

16.4 FINDING SOLUTIONS: BRIDGING THE GAP BETWEEN THE PARTIES

Experiences in implementing this framework have shown how scientific issues act as a wall that divides scientists from managers and industry representatives. The main problem is that tools, such as stock assessment, risk analysis and decision analysis, are highly technical and usually cryptic to non-specialists. Peterman [19] points out that it is hard to convey assumptions, results and implications effectively to people who are not actually involved in the analyses. In spite of this, the analysis of experiences has shown that a bridge can be built between scientists, stakeholders and managers by adding a bit of good will and imagination. For instance, in the beginning of the development of the South African anchovy MP, effort was made in assisting industry members to understand the stochastic aspects of the results. To achieve this goal, care was taken to emphasise underlying assumptions and sources of uncertainty [2].

Stock variability in short lived species is usually difficult to communicate to managers and stakeholders because it requires the use of distribution statistics, which are difficult for non-scientists to assimilate. This was reported in the case of the South African pelagic OMP. In order to address this problem, scientists developed computer simulation games to familiarise industry and decision-makers with the range of possible outcomes under different management procedures [14]. Francis and Shotton [13] recommend that the
statistics used to evaluate the performance of candidate management procedures should be intelligible to managers and stakeholders. For example, although quantities such as standard deviation or coefficients of variation of catch limits have been used in some studies, it is rare for non-specialists (for example, managers and members of the industry) to find these statistics particularly useful.

The presentation of simulation results is one of the means of enhancing stakeholders’ understanding of the performance of alternative strategies. Since the output produced by a comprehensive simulation can be quite overwhelming, special effort should be made in presenting the results of these simulations in an understandable way. In this situation, a distinction shall be made between presenting results to scientists or presenting results to managers and stakeholders. In the first case, scientists will find information that enables understanding of how results were generated and performance statistics more useful and relevant. Therefore, tables and numerical results may be more appropriate to this audience. In the second case, the presentation of the simulation should promote communication and understanding. Thus, graphs should illustrate variability by a bundle of trajectories and risks by cumulated distribution. Other tools, such as radar plots, traffic lights and even animations, should be considered [28].

Smith et al. [13] report that, in Australia, the communication gap has been narrowed by the incorporation into the process of a scientist working for the industry, who acts as a catalyst for effective communication, backs up industry views with science, revisits the work of the other scientists and communicates their findings to the stakeholders in an understandable way. Moreover, this industry scientist has helped to establish and run the industry’s monitoring and sampling programme. Industry funded scientists guarantee the transparency of the scientific process. Workshops and seminars organised by the scientific community in which stakeholders have the opportunity for active participation are another means by which to enhance communication and understanding between the parties. Smith et al. [3] consider that there is much to be done regarding understanding of the concepts, methods and terminology. Enhancing understanding of these factors will prevent confusion and the temptation to change recommendations and agreements at particular levels during the implementation process.

To sum up, during the process significant progress was reached towards overcoming both mistrust between fishers and fisheries managers and researchers as well as factional differences within the fishing industry [3]. This success can be attributed to the co-management approach used in Australia; the application of MSE would have been difficult under another management regime. Cochrane et al. [2] report that in South Africa the use of management procedures have greatly improved communication with the industry and brought meaningful knowledge and participation to the process. These experiences support the argument of Jennings et al. [31] who point out that co-management is an excellent means for promoting understanding between fishers, scientists and managers.

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REFERENCES


Section 4

SUMMARY AND CONCLUSIONS
Chapter 17

The Role of Science within Modern Management Processes with the Development of Model-Based Evaluation Tools

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17.1 INTRODUCTION

It is widely accepted that the exploitation of the sea, together with environmental change, is causing substantial alteration in the marine ecosystem. These changes are taking place at a pace that overcomes the ability to manage living resources, as it challenges our capacity to generate the necessary knowledge for effective management.

The sustainability of fisheries and the marine ecosystem is at stake. At the global level, the Johannesburg declaration [1], widely accepted worldwide, implies a strong focus on objectives relating to sustainability, long-term yield and maintenance of ecosystem health. Despite this, examples of failures in the management of fisheries come from all parts of the globe. In particular, in Europe a politically complex, autocratic management system has given rise to a general perception of failure. Advice provision is difficult because operational management objectives, deemed to be set by a central authority, are not explicit; and even implicit objectives are not accepted by all the different stakeholders.

In this top-down management system, the knowledge base is oriented towards resource sustainability, as decided by key scientific advisory organisations, with no economic or social considerations included in the generation of formal scientific advice. However, in its later stages, the decision-making process tends to be highly influenced by economic, social and political considerations. In practice, this has meant that, although long-term sustainability has always been an objective of managers worldwide, a focus on yield optimisation and short-term considerations have determined decision-making outcomes, leading to overexploitation. In addition, the knowledge base within this system is opaque to the industry and other stakeholders: they do not participate in it and the interpretation of outcomes is divergent among the different actors. All this reduces the reliability of the system, weakens the legitimacy of scientific advice and of the management system as a whole, and increases non-compliance with regulations, which, in turn, has a deteriorating effect on the quality of the information base for the generation of knowledge.
New forms of management are needed to reverse this situation. Launched at the beginning of the twenty-first century, the reforms currently being implemented within the European Common Fisheries Policy (CFP) are a step towards overcoming these problems and to constructing the basis of a more efficient, integrated management system, with shared objectives, an adequate knowledge basis and operational procedures, participatory decision-making, and good levels of compliance (see Chapter 8).

As pointed out in the introductory chapter (Chapter 1), the review of the knowledge basis for fisheries management accomplished in this book is aimed at ascertaining the context in which tools for the evaluation of management options are to be used, the problems they are to assist in solving, and the knowledge they should communicate. As a corollary of this effort, we will outline in the following pages what the knowledge base and the role of science should be in modern fisheries management systems.

17.2 MEETING NEW MANAGEMENT CHALLENGES

New ideas have been developed to meet the challenges of natural resource management, and of fisheries management in particular. A more integrated, holistic approach to fisheries management has been rapidly emerging in the last decade. The eruption of the precautionary approach (PA) and its general adoption in fisheries management during the 1990s changed the focus of management towards long-term sustainability, taking into account the inherent uncertainty in all aspects of fishery systems. The burden of proof was placed upon the fisheries themselves to demonstrate that they are not putting fish populations, the ecosystem, and even the safety of the resource for new generations, at risk [2]. In this context, social pressure demands that clear objectives are set concerning both the sustainability of the resource and the economic and social viability of the industry in the long-term.

Participation of stakeholders other than managers and scientists in the establishment of entire management strategies—for example in co-management experiences—is becoming common practice. Of course, an integrated system does raise conflicts of interest among stakeholders groups, but the process will become more transparent and will gain in terms of legitimacy, should conflicts be overcome. When the views of stakeholders are introduced into the problem-solving process, the scientific formulation of issues can be elaborated with richer inputs but with new complexities (see Chapter 9).

The demand for better management requires clearly specified policies and equally clearly specified rules to achieve them. Management approaches have been devised that enable the evaluation of the performance of alternative strategies to reach pre-agreed objectives. This evaluation can be performed by trial and error experience or by experimental work; however, such scientific methods are, respectively, dangerous and impractical when applied to natural resource management. An alternative way, linked to the fast development of information technology capabilities, is the use of computer intensive modelling techniques to simulate alternative management scenarios as an aid to decision-making. The comparison of the outcomes of simulations allow for the evaluation of trade-offs between the performance of different strategies relative to the objectives and to test the robustness of the system to the various sources of uncertainty. Such an approach
confronts users with choices to explore rather than answers to accept or reject. It allows uncertainty to take the centre stage in the discussion among stakeholders regarding how an adaptive management process should be steered. Chapters 15 and 16, respectively, describe the use and analyse the constraints of management procedures (OMPs) and management strategy evaluation (MSE) frameworks, which are simulation tools that allow management procedures to be developed in line with the precautionary approach.

The management procedure approach is to become more widespread in Europe because the current fisheries policy in the European Union (EU) [3] recommends that multi-annual recovery plans should be used for depleted stocks. Changing from yearly to multi-annual management arrangements will enable the eradication, or at least the reduction, of one of the more flawed features of the CFP management system: the annual fight for Total Allowable Catch (TAC) setting and quota allocation during the December meeting of the Council of Fisheries Ministers. Multi-annual management is based on definite harvest control rules (HCRs) designed to reach pre-agreed management objectives. In this way, current implicit HCRs, focused on short-term quota availability [4], are to be replaced with the explicit HCRs leading to long-term management (biological and socio-economic) sustainability objectives. This will reduce the influence of short-term demands for unsustainable harvesting, while simultaneously ensuring both greater stability in exploitation rates and greater flexibility for management negotiations as management decisions are addressed over a longer time horizon. Multi-annual advice switches short-term interests for strategic long-term perspectives, and reduces the burdens on the knowledge provision system. It clarifies the management process and enhances the commitment of stakeholders to the plan. Uncertainty and risk are managed explicitly and short-term gains/losses are set against long-term losses/gains both biologically (population sustainability) and socio-economically (fleets and industry).

Chapter 16 points to economic and knowledge constraints that can impede the management procedure approach from replacing conventional management in a global context. However, despite these constraints, such systems are expected to play a key role in managing fisheries in the near future [5]. Although the costs of their implementation and maintenance may be too high for governments to bear, it is possible for the burden of implementation to be shared with, or even entirely borne by, industry. Such a cost redistribution can also lead to more collaborative ways of producing knowledge.

Finally, an appropriate control and enforcement system is a cornerstone for ensuring the sustainable exploitation of resources. Understanding why fishers do not comply with regulations is essential for the development of appropriate fisheries management systems, and is, thus, an important part of the fisheries knowledge base itself (see Chapter 14 for a discussion on the basis and impacts of non-compliant behaviour in fisheries management). Changes in the management system can favour compliance. Incentives to comply might be improved through the increased use of rights-based management systems. Further, it should be noted that compliance and enforcement are simpler to achieve under effort regulation systems than within output regulation systems because it is easier to observe the activity of vessels than to monitor catch and output (Chapter 7). Participation of stakeholders in key management decisions, or ‘co-management’, would bring greater transparency and legitimacy to the system, building up the necessary confidence in management measures to achieve an increase in compliance.
17.3 THE KNOWLEDGE BASE FOR MODERN MANAGEMENT

The knowledge base includes biological information about fish stocks, economic information about fisheries, ecological information on the functioning of the ecosystem and the influences of natural changes and anthropogenic activities on it, and socio-political information about the requirements of effective fisheries governance. The knowledge production structure should reflect the management strategy to be adopted, while, at the same time, the management strategy should take into account both levels of uncertainty in the current knowledge base and the implications of management actions for future knowledge gathering. Knowledge-based processes should be established to ensure that management is effective and meets the stated objectives. Performance criteria should be decided upon in conjunction with decisions regarding objectives (cost-effectiveness; robustness) in order to enable the evaluation of management outcomes. The challenge is to ensure that the best available knowledge, including information about the relative certainty of that knowledge, is made available, in an adequate form, to the decision-making process to inform and assist decisions and communicate complex insights effectively to the increasing number of stakeholders involved in these decisions.

Participation of stakeholders in the creation of the knowledge base is also often critical and can be facilitated by cooperative scientific activities, subjecting stakeholder claims to transparent evaluation, and increasing the use of frameworks for evaluation of management measures in negotiations over management strategies. As discussed extensively in Chapters 4 and 9, collaboration around the knowledge base takes many forms: from stakeholders acting as little more than research assistants, to stakeholders acting as full collaborators.

17.3.1 Decision-making with uncertain knowledge

Fisheries stock assessments are conducted to evaluate the consequences of alternative management actions: to conduct a decision analysis. Regardless of the reliability of the information base, fisheries science has tried to construct increasingly complex methods and models to meet this requirement. Uncertainties in the information base include the inability to predict fish population dynamics, stock productivity, food-chain interactions and environmental and anthropogenic effects. Therefore, different interpretations of assessment outcomes are possible, a situation which leads to competing hypotheses about the dynamics and state of a fishery. The consequences of management actions may differ depending on which hypothesis is true. New methods are needed to enable fisheries stock assessments to deal with uncertainty [6], both for the cases where the calculation of risk, and hence probabilities, is feasible and for those cases where alternatives must be assessed by qualitative means.

The uncertainty and risk resulting from the limitations in fisheries management systems and scientific information, as well as natural variability, are recognised and should be taken into account by adopting more precautionary management strategies. Chapter 15 points out that better management is not necessarily going to be achieved by improving stock assessments because, even with a perfect assessment, stocks may collapse at apparently safe fishing levels.
The orientation of the European advisory system towards the setting of TACs, instead of a more holistic approach towards understanding the processes affecting the dynamics of fish populations, has contributed to the general lack of knowledge. Research effort should be devoted to ascertaining cause-effect relationships in critical fisheries processes, whether naturally or human-induced. The move from yearly assessments to multi-annual plans decreases management reliance on year-to-year scientific advice relating specifically to the health of commercial fish stocks. Within the context of a multi-annual approach, advice needs to focus on the consequences of different management strategies for the sustainability of the resource and for the viability of the fleet/fishery.

The knowledge basis for the PA framework is focused on stock productivity and aims to avoid recruitment overfishing [7]. The stock recruitment relationship (SRR) is a cornerstone in this context. However, uncertainty in SRR hampers the reliability of the PA advice. The SRR is a fisheries science dogma—there is an underlying assumption that the stocks do not compensate for reduced abundance by increasing productivity—which is hardly demonstrable by evidence and widely unaccepted by fishers. Uncertainty in this case comes from measurement (catch and discard levels; recruitment index) and process errors (lack of understanding of what drives the process, including the influence of external factors, such as regime shift or environmental change), making it very difficult to discern cause-effect relationships from their noisy backgrounds and leading to an uninformative lack of clear indicators in most cases. Uncertainty in SRR critically affects to the estimation of MSY-related biological reference points, although it is used extensively in legal text as a reference for sustainability.

17.3.2 Management option evaluation by scenario modelling

An evaluation framework can structure and communicate existing knowledge in terms of data and knowledge about processes, and enable exploration of options within specific management procedures, with an evaluation of trade-offs between various objectives and of the robustness of the options (to assumptions, and to model and data errors) and risks involved. Scenario modelling is an alternative way, compatible with PA, to set up such a framework by comparing the accomplishment of different performance indicators for alternative management options and testing its robustness to uncertainty (Chapter 15 for a review of management procedures evaluation frameworks). In this way, the focus is on robustness regardless of the degree of knowledge (i.e., uncertainty) about the underlying processes. The simulation framework allows for investigation of our ability to understand, monitor and control these systems.

These operating models should capture the plausible range of characteristics of the underlying dynamics, but not necessarily model their full complexity [8]. In general, they will be more complex than those used for standard stock assessments and, thus, will involve the development of management procedures that are more robust to a broad range of uncertainties. However, the models and rules used as part of the management procedure should be simpler than those used at present. A simpler assessment method able to pick up overall trends in the stocks may be sufficient when used within a suitable management framework.
Implementation errors and the effects of non-compliance appear as new elements of uncertainty that are seldom taken into account (Chapter 14). This undermines assessments and, therefore, advice. The effects of non-compliance can be incorporated into fisheries bio-economic models in order to estimate more realistic outcomes of management policies. But it is not straightforward to achieve this.

17.3.3 Towards an integrated management approach

In an integrated approach, the traditional knowledge base focused on the sustainability of the biological resource needs to incorporate knowledge from other fields relevant to management, including ecology, economics and sociology. Fishers’ knowledge should also be integrated with the process of advice generation and decision-making in order to improve the quality of the information base, give legitimacy to the process and increase compliance. Modelling and scenario building, as discussed in Chapter 13, provides a possible way to structure these complexities within the creation of long-term management strategies.

The system should be prepared to provide an adequate basis for the more complex demands of new management strategies. In Europe, data collection, assessment tools and form of advice support the setting of annual TACs for single stocks. But the system itself produces unreliable data: namely data from stocks subject to restrictive TACs. New management measures are needed to achieve objectives in an effective management system. As a result, there is a move away from the traditional single stock advice and towards fisheries advice, involving adding in ecosystem considerations and, eventually, advancing towards ecosystem-based advice. Output (TAC) management implementation, although defended in Europe by a substantial part of the industry during the discussion of the CFP reform, tends to be complemented by input (effort) based management. With regard to the knowledge base used for deciding management measures: while in output management systems it is necessary to have relatively precise knowledge and certainty about resource abundance; effort-based management systems are less dependent on precise yearly stock assessment estimates and have a reduced need for annual assessment (Chapter 7). However, an input-based system will still need reasonably precise stock assessments and forecasts to set an appropriate initial effort level and to respond to changes in efficiency.

According to recent reports from the Committee for Fisheries and Aquaculture (ACFA) of the EU [9], EU data collection programmes are to be improved and re-focused to meet the needs of new approaches to fisheries management (fleet- and area-based management, rather than fish stock-based) and to make possible the move towards the ecosystem approach to fisheries management (EAFM), including the already achieved prioritisation of fishing effects and associated indicators to support the integration of environmental factors into the CFP. In addition, the implementation of a more regional dimension to fisheries management is expected to enhance stakeholder participation in data collection. A first set of data requirements to support the environmental integration process and the development of an ecosystem approach to management has already been established (including fishing effort and spatial distribution, assemblage composition, diet composition, key species, by-catch of marine mammals, turtles and seabirds, physical-chemical conditions, and seabed factors).
17.3.4 Constraints for introducing modern management procedures

Political constraints, such as the relative stability principle, act against innovations with respect to the traditional management system. The principal management instrument of the CFP, from its origin, has been the annual setting of TACs and the allocation of national quotas according to the principle of relative stability. Despite the fact that this process has been largely unsuccessful in controlling exploitation rates and that recovery plans and management plans are now being introduced, the annual TAC and quota-setting process remains the primary focus for fishery managers [10]. The need for TAC advice has been driven by the long-held principle of ‘relative stability’, which determines the distribution of relative fishing opportunities between EU Member States. However, the TAC itself is an impediment to the development of a cost-effective advisory (management) system, as it brings with it a series of drawbacks, including deterioration of catch statistics (landings, discards, bycatch) and consequent unreliability of stock predictions, and an inability to limit fishing mortality according to the set TAC. Concerning mixed fisheries, the advice system is hampered by deficient data quality and fishers’ behaviour. This is, in part, because of the lack of flexibility the relative stability principle gives to the system, which does not allow for restrictions/changes in the effort of individual fleets. On the other hand, fisheries advice increases transparency by making more explicit the outcomes for each fleet segment. There is a clear need for policy inputs in this kind of advice/system concerning priorities of fleet/species (Chapter 11). The success of such a system is clearly influenced by fishers’ reactions to the relative availability of quota between species/fleets.

The main problems concerning effort advice, in addition to those problems already mentioned with regard to the relative stability principle, relate to the comparability of effort units among fleet components and to quantifying technological creep (Chapter 7). Renewed focus has been placed on the limitation of fishing effort as a way to limit pressure on stocks—this is specifically within the framework of multi-annual recovery or management plans [3] but can be discerned as a clear trend within the CFP’s management measures.

The emergence of the EAFM will bring to the management system an increase in levels of uncertainty due to the lack of understanding of how ecosystems respond to fishing (Chapter 12). EAFM necessitates the development of scientific and institutional approaches for dealing with this inevitable increase in uncertainty without incurring a loss of legitimacy. The ecosystem approach needs additional monitoring, scientific investigations and complicated models and reference points, but, as economic resources are limited, it is important to develop cost effective performance indicators and Ecological Quality Objectives (EQOs) that could provide the basis for a knowledge base for managing ecological effects. However, the definition of ecosystem reference points and key indicators is at a very early stage in its development. Knowledge on the effects of Marine Protected Areas (MPAs), and technical measures compatible with fishing practices that are sustainable for the ecosystem are also to be developed. A probable consequence of the EAFM is that research survey objectives must be shifted from single species (or multi-species) evaluations, aimed at providing abundance indexes of target populations, to ecosystem evaluations, covering the monitoring of a broad set of ecological parameters and processes.
Developments in the social science of fisheries, and its knowledge base, are essential to enable managers to learn how to understand the drivers of the management process and how to address the ongoing problems that arise from conflicting objectives and perceptions of problems. For a long time, perhaps, the central social science message regarding management and its knowledge base has been a plea to avoid thinking about what is an essentially political process as a technical one, which is finished when an answer is generated of how to reach a particular “clear objective”. As discussed in the introductory chapter (Chapter 1), the approach taken by social science to the problem of managing natural resources, such as fisheries, has shifted away from a focus on institutional design and towards the analysis of institutional processes responding to constant changes in the social and natural management environment. This has meant that social science questions about the formation and use of the management knowledge base have shifted from the periphery of the discussion to its centre.

The constraints on manpower and time affecting existing actors are compromising the advisory system and are proving to be major impediments to improving the quality of assessments and advice. Indeed, the European Commission itself states that: “the core of the problem is that there are simply not enough scientists to provide the analysis and advice which the Community needs to operate the common fisheries policy” [11]. However, in the light of the knowledge base required for new management trends, it is clear that there will not be a lesser need for qualified scientific advice. Rather, new demands on advice relating to the ecosystem approach and management strategy evaluations might very well put more demands on an already overstretched system. New demands are already emerging for scientists within the multi-stakeholder management process, which requires them to move beyond providing objective information to facilitating transparency around claims to knowledge in the policy-setting process.

A lack of technical stock assessment skills, and of modelling skills in general, can constrain management. The increase in the complexity of the management system and tools magnifies existing problems because the operating models are often markedly more complex than the models underlying most stock assessments (Chapter 16). Moreover, the implementation of these approaches requires the development of special software, the use of powerful computers, training of staff, hiring of consultants, improvement of the data collection system, and organisation of discussion and informative sessions. The primary objective of the EFIMAS project is to create such software and provide training for its use. The ICES Study Group on Management Strategies [4] reviewed the software available for evaluation of management measures (in particular, HCRs). For example, a generic framework FLR (Fisheries Science in R) [12] has now been developed by several European projects. Use of these packages should make the extension of the scenario simulation approach to new fisheries possible.

17.4 THE ROLE OF SCIENCE

A key requirement for effective management is the production of information upon which adequate decisions can be based. It is the role of science and scientists to provide the best available information on the state of the resources and their likely response to
management measures, notwithstanding the legal responsibilities of management institutions and industry for obtaining the information base for assessment: catch and effort data.

The precautionary approach calls for a management system based on the best scientific evidence taking into account uncertainty. The scientific method provides knowledge-based information and its primary strength is that it provides a way for people to explain in the most transparent way possible how they know what they know. Given the uncertainty of the information base, fisheries scientists are often faced with the dilemma of making decisions based on insufficient information. As our knowledge base increases we often learn that what was previously considered best practice was not the most appropriate solution. This modifying of solutions is unavoidable in science, as well as in other practices, where the objective is to find truth. However, the management strategies evaluation approach turns the focus not to selecting the truth but to testing the robustness of the management strategy to uncertainty. And it is adaptive in itself, reckoning that our understanding of natural processes is limited and making it possible to feed back into the system new understandings gained during implementation. Even more than objectivity, which always remains an elusive claim in a complex society, transparency provides the scientific basis for the legitimacy of management decisions (as discussed in Chapter 13).

17.4.1 Science for scenario modelling

Management Procedures (MP) are powerful tools, which are firmly rooted in the existing knowledge base (Chapter 16). They enable a systematic analysis of the knowledge and control requirements of alternative management instruments. In addition, these MP analyses facilitate the evaluation of research needs for the improvement of management. Thus, analysis can be directed towards testing the robustness of alternative assessment methods and management strategies for alternative plausible hypotheses about which there is little prior knowledge, and to identify where improved knowledge of stock dynamics is needed. This cannot work equally well for all stocks, and the results such a system can give should be estimated according to its features regarding robustness and the predictability of the main parameters and indicators. Users of MPs also need to be adaptable and vigilant to change: quick reaction is important when assumptions are no longer valid or critical changes are perceived.

The effectiveness of the evaluation framework in informing stakeholders in the decision-making process and assisting their exploration of management options should partly be evaluated in processes where stakeholders use the framework for exploration. The outcomes of these evaluations should be used iteratively to improve the capabilities of the evaluation framework to inform and assist decision-making processes further. Management simulations should be used as part of a dynamic process involving dialogue between scientists, managers and stakeholders, where the role of science is both to evaluate proposed strategies and to provide advice regarding which kinds of strategies can be worth considering according to their performance in meeting agreed objectives. In the future, evaluation frameworks can structure a systematic requirement for transparency of knowledge claims within negotiations over management strategies.
The complexity of models and of evaluation frameworks is a problem for non-specialists. The growth of complexity and the internal assumptions of these scientific tools make it difficult for stakeholders to understand what drives many assessments, thereby reducing their accessibility. As few people are able to understand and apply them effectively such systems are seldom understood by managers and industry, in spite of their active participation. The effective communication of science-based policy is needed to ensure that evaluation tools attract positive stakeholder perceptions. Developing understanding between scientists, managers and stakeholders is a key factor in achieving this. The field of cognitive psychology and social and management sciences offer the best opportunity to find the means to ease communication between the diverse groups interested and involved in fisheries and its management.

17.4.2 Stakeholder participation

Participation of stakeholders has been very limited in the European management system. This is the case regarding both the generation of knowledge—data collection, diagnostics of the state of the resources, generation of advice and monitoring of resources—and the decision-making process, including the definition of management objectives. In terms of the knowledge base and the scientific process, closer working relationships between scientists and fishermen are essential. Fishers’ knowledge can improve the knowledge base in aspects related to day-to-day fishers’ experience, such as the location of nursery and spawning areas and the migration patterns of key commercial species. In other fisheries, there are successful examples of industry surveys carried out following scientific methodologies (for example, Iceland Commercial Surveys).

The recent introduction of Regional Advisory Councils (RACs) in the EU may be a step towards improving collaboration between stakeholders and scientists (see Chapter 8 and Chapter 9 for an extensive discussion on RACs and stakeholder participation). The main task of RACs is to advise the Commission and/or Member States on matters relating to fisheries management in respect of designated sea areas or fishing zones. However, the way they are set up does not explicitly ensure either that the system will be more participative or that the generation of scientific advice will be shared by stakeholders. As they have been devised, RACs are a means to ease communication among different stakeholders: namely industry, conservationists and scientists, with a secondary participation of other interested groups. However, thus far, these regional fora are mostly driven by representatives of the fishing industry and it is not currently envisioned by the Commission that they will share in the tasks involved in generating scientific advice. Further, although actors involved in both industry and in the scientific community have expressed an interest in expanding the RACs’ role in generating and validating the knowledge base, and while the basic structure of the scientific advice system is currently under intense discussion, for the foreseeable future ICES will remain at the centre of the advisory system. ICES is committed to “providing relevant scientists to attend” the meetings of RACs “at which a dialogue between scientists, fishing sector and other stakeholders will be promoted”. ICES intends to work constructively and interactively with the new RACs. Early in 2006, RACs will be invited to discuss the format and extent of such interaction (excerpts from the Presentation by David Griffith, ICES General
Secretary in the constitution meeting of the NWW RAC, Dublin, 30 September 2005). As a result, it is not clear whether current efforts to integrate fishers’ knowledge into ICES’ processes are going to work.

On the other hand, for the time being, only discussion of ACFM advice for commercial stocks is included on the RACs’ agendas, with little consideration being given to data and assessment procedures. As a result, one may see these first steps as a mere enhancement of communication between scientists and industry. Given their predominance within the RACs, only the industry (with potential support from scientists) can drive RACs towards a more participative scientific set-up, by way of increasing their own scientific activities—including data collection and incorporating fishers’ and other stakeholders’ knowledge in assessments—and supporting the production of scientific advice, while also helping to determine its form and use.

Technical measures and socio-economic evaluations are issues that RACs wish to emphasise alongside the biologically-driven scientific advice prevalent in the European fisheries management system. In this context, the use in RACs of Management Strategies Evaluation Frameworks, such as the one being developed by EFIMAS, can enhance participation and enable European fisheries to advance towards a truly shared management system. However, no EU funds have been allocated for any (regional) scientific studies that the RACs might wish to initiate. It will become clear in the future whether or not RACs are the embryo of co-operative decision-making procedures within the CFP. If this is to be the outcome, there would need to be profound changes in the primary legal foundations of the CFP, under the terms of the Treaties of the EU, as these foundations have been argued as the reason not to give decision-making powers to the RACs.

17.5 CONCLUSION

Extensive literature shows that the effective communication of advice is critical for its acceptance by all actors in the decision-making process (both Chapter 13 and Chapter 16 review this topic). Effort should be made to adapt the way science communicates complex results to their non-specialist audiences. Scientists have been accused of being insufficiently transparent in the development and delivery of their advice. This lack of transparency undermines the credibility of such advice among stakeholders, who may very well feel that uncertainties and other important factors are hidden from them. Procedures for generating scientific knowledge and constructing advice should thus be made more transparent and the presentation of results should promote communication and shared understanding.

European fisheries management is moving towards a long-term, adaptive approach that takes into consideration a broad set of ecological issues, while involving multiple stakeholders. Because of this, the role of science in the management process will undergo profound changes. Scientists will be asked to take on more complex roles within management, well beyond being the providers of objective knowledge to support the decision-making of others. In this volume we have tried to outline the major issues that must be addressed and the innovative ways that scientists are finding to address them. They have much to contribute because of their experience with the generation of knowledge.
through transparent methods and processes. Accountability through transparency and making use of the knowledge and commitment of different stakeholders will make possible a management system that can adapt to complexity. The methods we have reviewed in this book for the analysis and evaluation of management alternatives are critical tools that scientists can use to play these complex new roles in a changing management system.

ACKNOWLEDGEMENTS

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### Acronyms

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<th>Acronym</th>
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<td>ABARE</td>
<td>Australian Bureau of Agricultural and Resource Economics</td>
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<td>Annual Catch Entitlement</td>
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<td>African, Caribbean and Pacific</td>
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<td>Catch Per Unit of Effort</td>
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<td>Community Support Framework</td>
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### Acronyms

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<td>Driving forces, Pressure, State, Impact, Response</td>
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<td>Driver Pressure State Response</td>
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<td>Distant Water Fishery Nation</td>
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<td>EA</td>
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<td>ICES</td>
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<td>Institute for European Environmental Policy</td>
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