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Introduction

NILS BUNNEFELD, EMILY NICHOLSON AND E.J. MILNER-GULLAND

1.1 THE NEED FOR THIS BOOK

Making decisions about the management and conservation of nature is complex and consequently difficult. The complexity stems from the many competing pressures on natural systems, with their opportunities and benefits for different groups of people, set within a constantly varying social and ecological environment. However, there are also opportunities for better decision-making, leading to better outcomes for all sides. This book showcases one such set of opportunities – the benefits of taking a structured, participatory, model-based approach to decision-making for biodiversity conservation.

The largely unrealised potential of using this approach to making decisions about wildlife management became very clear to N.B. and E.J.M.G. when we worked on an endangered antelope endemic to Ethiopia, the mountain nyala (*Tragelaphus buxtoni*). When we got involved in the project in 2010, the total population was estimated at just less than 4,000 individuals (Atickem et al. 2011; Bunnefeld et al. 2013). The pressures on this antelope are high because of a combination of hunting, habitat loss and poaching. The situation was complex due to high uncertainty about the population size because monitoring was limited and the impacts of habitat loss and poaching were unknown. The question to which the Ethiopian Wildlife Authority wanted an answer was how to set a sustainable quota that increases their income, while also providing benefits to local communities. The plan was to reinvest the funds into monitoring, habitat conservation and livelihood support for local people. We used a management strategy evaluation-type simulation model that incorporated the dynamics and uncertainties mentioned earlier (Bunnefeld et al. 2013) to find answers to this question. However, funding stopped before we could start the process of implementing a decision-making process, and the situation is currently unchanged, despite a pathway to better management now being available thanks to our collaborative research project. This failure to translate research into impact sparked our interest in finding examples where such a translation had taken place, and understanding the factors enabling it to happen. There are surprisingly few successful case studies; some of the best are presented in this book.

There is currently no end in sight for the present biodiversity crisis, or even a road map for slowing down current rapid biodiversity loss (Venter et al. 2016). Biodiversity loss is important for society at large because of the complex relationships between biodiversity conservation, food security and human well-being, including both synergies and trade-offs (Mace et al. 2012). Given the occurrence of environmental and climate change, and a growing human population to feed, human well-being will rely on better decisions to turn potential synergies between biodiversity conservation and human advancement into real-world opportunities for a positive change. The UN's Sustainable Development Goals (SDGs) define the way governments and businesses should set development priorities over the next thirty years (Terama et al. 2015). However, the prospects for real change are limited, if the evidence to date is anything to go by. Tittensor et al. (2014) suggest how the UN Convention on Biological Diversity's (CBD) Aichi targets are not producing the desired outcomes. What is certain is that an approach that is narrowly focussed on either human development or environmental protection cannot deliver sustainable solutions to managing the complex and uncertain social-ecological interactions and feedbacks, which constitute people's relationships with nature (Larrosa et al. 2016). With this in mind, this book brings together authors from a range of disciplines to reflect on their experiences, successful and less so, in effecting real-world change on the ground. Their experiences point to a new way to make decisions for sustainable resource management and biodiversity conservation, which may improve outcomes for both humans and nature.

1.2 A SHORT HISTORY OF QUANTITATIVE APPROACHES TO BIODIVERSITY CONSERVATION

Humans have managed and manipulated ecosystems for their own benefit for millennia, typically using conceptual models as a basis for decisions. There was an early realisation that resources are limited and that this has implications for the viability of humanity (Malthus 1798). In the 1970s, the tragedy of the commons was the prevailing theory explaining the inevitable overexploitation of natural resources (Hardin 1968), harming both the state of the natural resource and eventually people themselves through a shortage in the resource they depend upon or enjoy. The tragedy of the commons theory is based on an open access system where increased exploitation benefits a single person, whereas the costs are shared among all those using the resource (such as sheep grazing by different farmers on land over which they have no ownership). Later critiques highlighted that open access is only one of a number of potential land tenure situations; others, such as communal ownership, are more amenable to management (Ostrom 1990).

Conceptual models such as the tragedy of the commons were formalised as quantitative models to support decision-making about managing natural resources. Early examples in natural resource management (NRM) include the use of models to set sustainable fishing harvests (Gordon 1954; Beverton and Holt 1957). Such models stemmed from advances in ecology and mathematical modelling (and later economics), and predominantly found application in fisheries, forestry, agriculture and harvesting wildlife in the 1950s, based on the relationship between the rate of replenishment and growth of a resource (such as an animal or plant species) and the off-take of this resource. This was formalised into the maximum sustainable yield (MSY) – the point at which the maximum number of individuals can be taken from a population without causing a decline in numbers. Classical MSY is, however, based on strong assumptions, such as that the environment is deterministic, all individuals in a population can be represented in a single value for population size (rather than structured by e.g. age, sex or spatial location) and under the simplest formulation of density dependence (logistic growth), that density dependence operates symmetrically so that MSY is found at half of the carrying capacity (the maximum population size; Clark 1990).

In the 1990s, there was a realisation that stochastic events, such as year-on-year changes in weather and demographic variability, lead to fluctuations in population size and growth rate, increasing the chance of overexploitation and even extirpation of a local population if hunting is too heavy. Models accounting for this uncertainty led to recommended harvesting limits being related to the degree of variability in population dynamics (Lande et al. 2003). However, these models also make assumptions which are not seen in reality, such as that managers have complete understanding of the system that they manage (monitoring uncertainty) and that harvesters accurately follow agreed-upon harvest goals (implementation uncertainty); both have been relaxed in more recent models (Fryxell et al. 2010; Fulton et al. 2011b; Bunnefeld et al. 2013). These models also focus on single-species harvesting, assuming that the interactions between a given species and other components of the ecosystem are not key determinants of sustainability.

A different stream of thinking about human relationships with nature promoted the protection of natural resources and land, rather than NRM and sustainable use of wildlife. Protection of land (and later aquatic environments) from human exploitation in order to halt ongoing loss of habitat and biodiversity initiated the modern Western conceptualisation of conservation in the nineteenth century, with a strong focus on human exclusion, through the establishment of protected areas (PAs). This included the establishment of national parks in the USA – for example Yellowstone National Park in 1872 (Cross et al. 2012) – and in colonial Africa (Adams 2004). Although this conceptualisation is usually credited as originating from the USA, the theory and practice of separation of humanity from nature in order to better protect nature is widespread and ancient, manifesting itself, for example, in Indian sacred groves and Russian *zapovedniks* (Bhagwat and Rutte 2006; Degteva et al. 2015). The number of recorded PAs increased after World War II, especially in Africa and Latin America, and doubled globally during the 1970s. By 2014, around 209,000 PAs existed covering 15.4 per cent of terrestrial and inland water areas and 3.4 per cent of the oceans (Juffe-Bignoli et al. 2014). During a similar timeframe, conservation societies were founded, starting in 1903 with Fauna and Flora International, which is considered to be the first society dedicated to the conservation of wildlife through saving habitat (Adams 2004). Conservation then was focussed on preservation, PAs and small population management of species considered as valuable, mostly by foreigners for aesthetic reasons, rather than by local communities for sustainable resource use (Caughley 1994).

This type of conservation has traditionally been rather separate from the fields of applied ecology and wildlife management, because conservation in its initial conception prevents any use of natural resources. In an effort to bring the two fields together, the International Union for Conservation of Nature (IUCN) General Assembly passed the Kinshasa Resolution on the Protection of Traditional Ways of Life in 1975, asking governments not to displace people from PAs. Also, the CBD, signed at the Rio Summit in 1992, recognised local people as having rights to use their resources. Integrated conservation and development projects (ICDP) were initiated in the 1990s to create revenues from conservation (i.e. non-extractive use) and to provide incentives for local people to engage in conservation activities and comply with conservation laws. ICDPs were seen as a realistic opportunity for a win-win situation between conservation of biodiversity and development for local people (Winkler 2011). However, many ICDPs failed, for a range of reasons, many of which came down to them being implemented by conservationists who didn't understand the complexities of development as a field, and therefore made mistakes in implementation (e.g. the popularity of 'alternative livelihoods' projects; Wright et al. 2016). More recently, conservation thinking is getting closer to NRM. For example, in the oceans, we see the integration of fisheries management with marine conservation (e.g. the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) listing sharks at the Conference of the Parties in 2013). Similarly, the realisation that there are important non-timber forest products coming out of tropical forests (such as bushmeat, medicinal products and honey) had led to more integration of forest conservation with sustainable use, particularly where poor rural people are dependent on these resources (Laurance et al. 2012).

The exclusion of people from areas which they previously had access to as a result of the fences and fines paradigm of early conservation, and the restrictions on natural resource use required of local people under wildlife management, have often resulted in conflicts, which are widely recognised as damaging to both wildlife and human livelihoods (Redpath et al. 2015). Managing human use of an area can mean trade-offs between different ecosystem services, such as provisioning services (farming and food production) and social and cultural services (biodiversity), but also win-win situations, for instance when biodiversity underpins subsistence harvesting or provides new avenues for income generation, such as ecotourism. Understanding and working to resolve these trade-offs is key to improving socio-economic and ecological sustainability (Daw et al. 2015). Researchers have so far mainly addressed the problem by focussing on documenting the benefits of wildlife to human livelihoods and well-being. However, their research fails to address the challenge that arises when stakeholders have competing views on how natural resources should be managed, from a local (e.g. wildlife harvests) to a global scale (e.g. Aichi targets). Such conflicts are likely to increase substantially in scope and scale due to the rate of current climate change and its uncertain effects on biodiversity and food security (White and Ward 2010; Mace et al. 2012). Traditionally, the models used to represent and understand resource management issues have ignored the roots and consequences of these stakeholder conflicts (Fulton et al. 2011c), but it is clear that we need new approaches for exposing and negotiating trade-offs in order to resolve conflicts between stakeholders.

Partly as a response to these conflicts and trade-offs, conservation and wildlife management have both moved recently (in parallel mostly) towards ecosystem-based thinking and understanding interconnectedness on the biological side. They have also both explored links between nature and human well-being (broadly defined; not just income), e.g. through the paradigm of ecosystem-based fisheries management on the NRM side (Daw et al. 2016), and ecosystem services on the conservation side (Mace et al. 2012). Both conservation and wildlife management have developed approaches for integrated management of social-ecological systems, but both are having limited success in the implementation of such frameworks. Part of the problem is the lack of communication between research and practice, which means that researchers aren't asking the right questions and practitioners aren't setting themselves up to learn (Pooley et al. 2014).

1.3 FRAMEWORKS FOR CONSERVATION DECISION-MAKING

Three broad frameworks for conservation decision-making have emerged within academic circles over the last two decades, largely in parallel, but all stemming from decision science approaches from economics and with many common elements: (1) decision theoretic approaches to conservation, including conservation planning (Shea et al. 1998; Possingham et al. 2000), which have developed recently into structured decision-making, used for example for endangered species management under climate change (Gregory et al. 2013); (2) adaptive management in weed control and hunting wildlife (Walters 1986); (3) management strategy evaluation in fisheries (Smith et al. 2008). Examples of all three of these approaches are found throughout this book, but the degree to which they have been used in practice to inform real-world decision-making is both variable and generally low.

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Edited by Nils Bunnefeld , Emily Nicholson , E. J. Milner-Gulland
Excerpt
More Information

1.3.1 Decision Theory and Its Application to Conservation Decision and Planning

In the late 1980s, decision theory emerged as the basis of a new approach to conservation biology. In particular, the 1980s saw the emergence of the new discipline of systematic conservation planning (Pressey and Nicholls 1989; Pressey et al. 1993), later termed spatial conservation prioritisation (SCP), which required planners to set clear targets and objectives, identify constraints (such as costs) and then optimise land use accordingly (Chapter 9).

Spatial conservation prioritisation has had substantial success in some countries in forming the basis for conservation decision-making; for example in South Africa, it forms the scientific backbone for land use planning by the provincial government (Lötter 2014; Cockburn et al. 2016). Decision theory, in its pure sense, has been less widely translated into practice, although there have been applications in Australia and New Zealand, where it has been used by government to prioritise action for threatened species (Joseph et al. 2009; Szabo et al. 2009). Applications of decision theory increasingly deal with uncertainty, e.g. info-gap approaches that aim to make least-worst decisions in conditions of high uncertainty (Ben-Haim 2006).

Structured decision-making (SDM) is a new phrase used for many examples of decision analysis within conservation. The steps are very similar to those of decision theory, as advocated by Shea et al. (1998) for a structured approach to problem-solving (Figure 1.1): defining the decision context and objectives, possible actions, a model (be it quantitative or qualitative) to project the consequences and impacts of the possible actions on the objectives, defining trade-offs, setting up monitoring, as well as acknowledging, describing and examining uncertainty, and using similar methods for making decisions and implementation. SDM has been used, for example, in papers outlining approaches to making decisions for threatened species conservation under climate change (Gregory et al. 2013).

1.3.2 Adaptive Management

The idea of adapting actions to the fluctuations and changes in ecological systems led in the 1980s to the concept of adaptive management (AM) – a process of continual learning by doing during the process of management (Holling 1978; Walters 1986; Keith et al. 2011). Adaptive management is distinguished from trial and error management by the

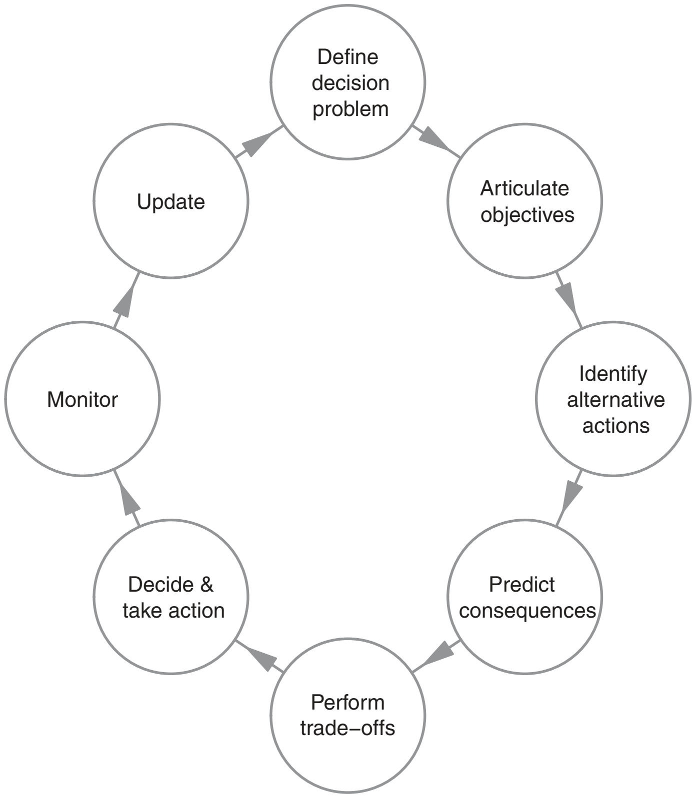

Figure 1.1 A representation of a structured decision-making approach.

prior intention to gain information while managing, in order to learn and improve, either passively through monitoring changes that transpire under management, or actively through setting management up as an experiment (Figure 1.2). Examples of passive AM include duck hunting in the USA (Nichols et al. 2007) and goose hunting in Scandinavia (Madsen and Williams 2012; Madsen et al. 2015) as well as the control of invasive plant species in Chapter 6.

While, in principle, the idea of adapting to change through monitoring and learning while carrying out management actions sounds plausible, many obstacles mean that the concept is well developed in theory by researchers but still not very much taken up by practitioners (Keith et al. 2011). One hurdle in AM is that the concept in its initial form did not include the responses of resource users to management into the equation. In fisheries, researchers recognised the importance

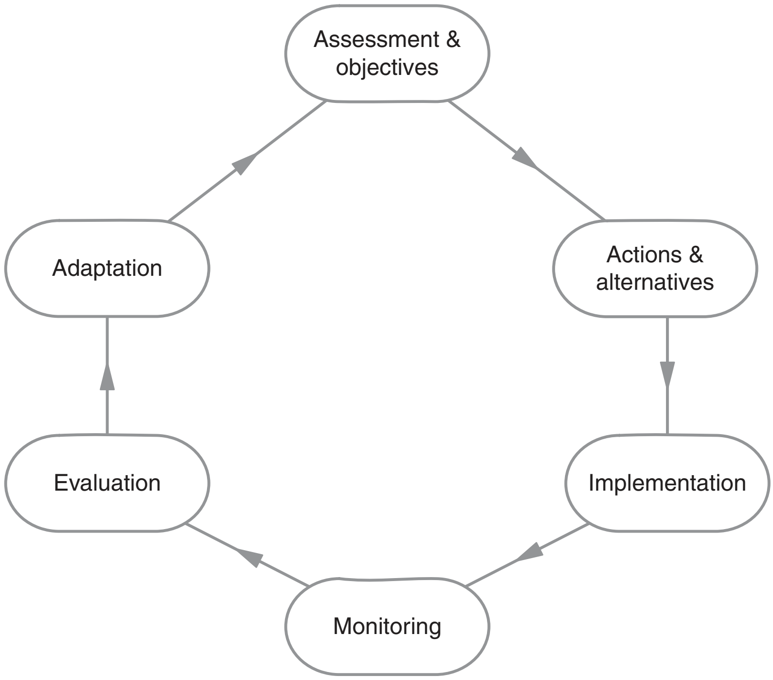

Figure 1.2 The adaptive management cycle: System assessment and setting objectives, setting up actions and possible alternatives, implementing the actions, monitoring the effects on the system, evaluating the system changes and updating knowledge and adapting objectives and actions based on these changes.

of understanding and adapting to people's responses to management in the early 2000s, but focussed their analyses on economic drivers of responses by industrial fishing fleets (Fulton et al. 2011c). Therefore, these insights are not always transferable to smaller scale operations where people are influenced by a range of social and cultural drivers beyond the profit motive (Clark 2006). Within academic conservation and NRM, there is a clear realisation that adaptive management is vital for improved decision-making, but in the real world, trial and error management, or management based on expert judgement and intuition, are still dominant, for a range of reasons set out by Keith et al. (2011).

1.3.3 Management Strategy Evaluation

Starting in the 1990s, there has been a much stronger integration of science and practice to improve management in fisheries, with the adoption of an approach called the management strategy evaluation (MSE). MSE originated from the Scientific Committee of the International Whaling Commission and was developed as a solution to the challenge of ensuring sustainable whale harvests (Punt & Donovan 2007). Eventually, the method was never used for whales, but the idea spread to fisheries management more widely and is now standard practice in many countries, including particularly Australia and South Africa (Punt et al. 2016). The strength of the MSE approach is that it takes into account three important components of wildlife management systems: (i) the ecological and social dynamics of the system itself, (ii) the observation/monitoring process with all its errors and biases and (iii) the assessment and decision-making processes of managers (Figure 1.3). MSE is especially useful when a system has quantifiable uncertainties, for example when the true population size of the resource population is not known, which is a common occurrence both in fisheries and terrestrial systems (Bunnefeld et al. 2013; Edwards et al. 2014; Punt et al. 2016). Uncertainty is too often ignored in the management of natural resources and conservation, whereas MSE puts it centre stage (Fulton et al. 2011b). Four main sources of uncertainty are addressed within the MSE framework (Bunnefeld et al. 2011):

(1) Monitoring uncertainty: managers do not directly observe the dynamics of the system, but do so only through monitoring;
(2) Structural uncertainty: there is uncertainty in how a specific system functions and responds to changes in the environment and management actions;
(3) Implementation uncertainty: management decisions are often only partially carried out by practitioners, e.g. harvest regulations are not always respected (Liberg et al. 2012) and pre-described quotas are not always filled (Knott et al. 2014);
(4) Process uncertainty: environmental stochasticity affecting natural resources makes it impossible to be sure of the correct parameters of the natural resource model.

One real world success which has been documented for MSE is the management of the Southern and Eastern Scalefish and Shark Fishery in Australia, a multi-species system comprised of many stakeholders with competing objectives, including local fishermen, environmental non-governmental organizations and governmental managers; this case study is described in Chapter 2. MSE was first implemented in