

Getting the Message Across

Biodiversity Science and Policy Interfaces – A Review

Biodiversity scientists should confront the political aspects of decision-making relating to biodiversity. Policy-relevant research entails careful assessment of social-ecological systems and possible trade-offs, taking into account the diversity of perspectives of the various stakeholders, being explicit about underlying framings and various policy options, and developing reflexive governance approaches.

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Abstract

This contribution analyses the difficulties of biodiversity policy-making and the role of science. It addresses biodiversity scientists' struggles to communicate the value of biodiversity to policy-makers, and the tensions between producing policy-relevant research and being perceived as too prescriptive. The author argues that it is important for biodiversity scientists to acknowledge and engage with the political aspects of biodiversity policies. A first step is to recognize the diversity of valuations and preferences among various stakeholders of different trade-offs, and the power relations between the stakeholders. A transparent presentation of a broad range of policy options, including an explicit description of the assumptions and models underpinning them, is a further requisite. Such presentations in turn require better understanding of the way institutions governing ecosystem (services) emerge and are established.

Keywords

biodiversity, ecosystem services, policy-making, transdisciplinary research

Despite the involvement of scientists in the *Convention on Biological Diversity (CBD)* and the organization of a number of high-profile scientific biodiversity assessments such as the *Millennium Ecosystem Assessment (MA)*, there is still a dominant narrative positing a huge gap between science and policy. Scientists complain that their warnings and urgent calls for action to conserve biodiversity do not seem to be taken seriously, and that scientific knowledge is ignored in the debates about agreements or policies, or “replaced by rhetoric” (Dietz and Stern 1998, p. 441). Some policy-makers indeed may ignore scientific knowledge that does not support their preferred policy options, while others complain they do not know what action to take given the uncertainty of some scientific conclusions. Yet, science is not undisputed either, as last year's public debates on Climategate showed, when scientists participating in the Intergovernmental Panel on Climate Change (IPCC) were accused of manipulating data to suit their own policy preferences.

Some of the problems with the biodiversity science-policy nexus may stem very well, according to Dietz and Stern (1998) as well as Koetz et al. (2009, 2008), from unreasonable expectations about how and how much science can contribute to wise decision-making. Often a technocratic approach is expected, both by researchers and policy-makers, in which science provides knowledge and information about the impacts of certain choices, and policy-makers use this information to design policies. However, in practice, the policy-making process is far from straightforward. Policy-makers have to weigh different socio-economic benefits and costs which are based on the values of biodiversity to society (Dietz and Stern 1998). To complicate matters, some of these valuations are subjective and emotional interpretations – both by policy-makers and their constituents (Bhattacharya et al. 2005). At the international level, trust – or a lack thereof – between states, especially in relation to capacities and the distribution of costs and benefits, also plays an important role in decision-making (Jessel 2012, Koetz et al. 2009, 2008). Each decision affects biodiversity, and effects take place at and across different scales, rendering both scientific predictions and decision-making complicated.

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This article analyses some recent research dealing with the difficulties of policy-making and the role of scientific knowledge and politics in this process. It starts with a discussion of how biodiversity scientists have struggled to demonstrate the value of biodiversity to policy-makers. It then continues to analyse some of the main challenges influencing science-policy interactions. These challenges relate both to the complexities of biodiversity as well as to those of the policy-making process itself – notably the politics of policy-making. The author argues that scientists should not ignore the political aspects of policy-making and suggests ways in which scientists can confront them.

Assessing Biodiversity

In 2005 the results of the *MA* were published. The reported conditions and trends relating to biodiversity were not very positive (*MA* 2005). The authors of the assessment admitted, however, that biodiversity still is an ambiguous concept that is difficult to operationalize and measure (Jessel 2012, Heink and Kowarik 2010). They posed that the knowledge on biodiversity shows a strong bias towards the species level – notably large mammals –, temperate systems, and biodiversity components used by people. They also noted that levels of uncertainty are quite high. Estimates of the total number of species on earth vary from five to 30 million, while only about two million species have been described. These widely diverging numbers result from the fact that little is known about, for instance, the total number of deep-sea organisms, fungi and micro-organisms. Nevertheless, extrapolating from the trends concerning known species, the *MA* concluded that in the past hundred years humans may have increased the species extinction rate by as much as three orders of magnitude. The main causes identified were habitat loss and modification, brought about by, for instance, conversion to crop land. Climate change was also pointed at as a possibly important driver of habitat loss and conversion, impacting negatively on biodiversity.

Although these alarming trends have started to become visible quite some time ago, scientists have long struggled to convince policy-makers to take their findings seriously and accordingly take action to conserve biodiversity. Mooney (2002) recalls the reactions to the 1995 *Geosphere-Biosphere Assessment* coordinated by the Scientific Committee on Problems of the Environment (SCOPE). Though signed by fifty-nine authors from different nations, policy-makers at the *CBD* questioned its legitimacy as they had not asked for it. More importantly, they felt it did not address the issues they directly encountered, as it only looked at the functioning of geospheres-biospheres, and not at their importance to society.

Assessing the value of biodiversity for humans since then has become a major effort for biodiversity science. A first step was to look closer at the relationship between biodiversity and ecosystems, which has become the major thrust in modern ecology, reflecting “a modern synthesis in which the study of biodiversity (e.g., distribution and abundance) is merged with the study of

ecosystem functioning (e.g., biochemical processes)” (Naeem et al. 2002, p. 3). A loss of biodiversity, including a loss of genetic resources, may result in a loss of productivity and a loss of buffering against ecological perturbation. It may also alter or harm ecosystem goods and services. However, different ecosystem processes may respond differently to this. This variability in combination with methodological disagreements resulted in sometimes fierce debates at the turn of the millennium about the importance of biodiversity to ecosystem functioning. These debates, Mooney (2002) argued, threatened to have negative effects on policy-making. They centred on correlational studies showing contradictory patterns of association between, for instance, plant diversity and production or other ecosystem processes, which raised doubts about the possibilities of generalizing results from biodiversity functioning experiments, and the relative value of such experiments when compared with real-life observation studies. Another point of debate addressed the relative importance of functional diversity versus species diversity (Naeem et al. 2002, p. 7, Loreau et al. 2001). Nevertheless, presently, there is consensus that biodiversity is of crucial importance to ecosystems and the services and goods they provide (*MA* 2005, Feld et al. 2010, Perrings et al. 2011), even though not all linkages are equally well-understood and sometimes difficult to quantify. Research is going on, focusing also on improved ways of assessing biodiversity (see, e.g., Feld et al. 2010, Heink and Kowarik 2010).

Valuing Biodiversity: Communicating the Need for Biodiversity Policies

A second important step was to find ways of assessing and communicating the value of biodiversity and (its contribution to) ecosystems to humans. This was and is no easy task, and many different approaches have been developed over time. Some involve the quantification of ecological risks associated with certain policy options (see Dietz and Stern 1998). Others involve attempts to assign monetary values to biodiversity and ecosystems (see, e.g., Brown 1987). Although the use of monetary values may seem a powerful way to address policy-makers, this approach triggered critique as well (Klie 2010, see Barkmann and Marggraf 2010 for a response). Not all benefits derived from biodiversity and ecosystems are easily translated into monetary values. Aesthetic appreciation or cultural values, for instance, are notoriously difficult to translate that way, though such values may very well influence policy decisions concerning what is deemed valuable to conserve (see, e.g., Schaich et al. 2010, Bhattacharya et al. 2005). Moreover, some aspects of biodiversity may contribute in ways that are less visible to policy-makers or wider audiences to services or goods that they do value. The ecosystem services concept, adopted by the *MA*, aims to address these questions by looking at benefits to humans in a broader sense – including non-consumptive and intangible benefits – and designing a framework that allows for defining ecosystem contributions more precisely and detailed (Ehrlich and Ehrlich 1981, Daily 1997, Mooney and Ehrlich 1997).

When an ecosystem directly or indirectly contributes toward meeting a human need or want, it provides a “service,” a contribution to human well-being, defined by the MA (2003, p. 73) as including “basic material needs for a good life, the experience of freedom, health, personal security, and good social relations” which together “provide the conditions for physical, social, psychological, and spiritual fulfilment.” The MA distinguishes the following ecosystem services:

- **provisioning services:** food, fresh water, wood and fibre, fuel, etc.,
- **regulating services:** climate, flood and disease regulation, water purification, etc.,
- **cultural services:** aesthetic, spiritual, educational, recreational experiences, etc.,
- **supporting services:** underlying processes such as nutrient cycling, soil formation, primary production.

These distinctions allowed for an extensive overview of the different ways humankind benefits from biodiversity and ecosystems. Yet, the MA framework still remains a utilitarian approach. This appears to be taken to extremes by some researchers who argue that distinction and categorization of ecosystem services allow for a more rigorous economic valuation of the different services (Daily et al. 2000, Boyd and Banzhaf 2007, De Groot et al. 2010), which would render the framework even more policy-relevant. Others, however, argue that this is quite difficult and perhaps undesirable for many of the important services – or biodiversity components, and have called for the development of alternative evaluation approaches (e. g., MA 2005, Klie 2010). Assigning economic values to the different ecosystem services entails a risk of rendering some of the services that had hitherto not been subjected to market regimes, marketable – which in turn may lead to privatization and hence the exclusion of certain users.

This can be especially problematic in developing countries, where access is often already difficult for poorer sections of the population. Attempts to conserve ecosystems and their services through privatization and marketing may very well clash with international declarations and agreements on human rights concerning the basic rights to food, shelter, and livelihoods (Duraiappah et al. 2005). These concerns are shared by some of those participating in *The Economics of Ecosystems and Biodiversity* programme (TEEB 2010, Ring et al. 2010). The latter have called for the integration of an ethical perspective in economic valuation, taking into account the vital significance of ecosystem services to the livelihoods of the poor, who often depend more heavily on environmental resources and who are hit hardest by the misuse of them. This would entail more encompassing ways of measuring human well-being – more in line with the broader definition used by the MA – through, for instance, the development of a “Gross Domestic Product (GDP) of the Poor”, which would highlight environmental resource-dependent sectors such as agriculture, animal husbandry, forestry and fishing, adding non-market benefits from these sectors and ecosystem services as well (Ring et al. 2010, p. 19).

Throughout the process of the assessment, the MA organizers tried to strengthen the links between the scientists and policy-makers. Representatives of some of the main conventions such as the *CBD, Ramsar (Convention on Wetlands of International Importance)*, as well as some other “user groups” such as indigenous peoples’ organizations and the business sector, were invited to the MA meetings and participated in the review panels. The MA has indeed managed to capture the interests of policy-makers around the globe, and stimulated further research directed at their needs (Larigauderie and Mooney 2010, Carpenter et al. 2009). Though that can certainly be referred to as a success, science-policy interfaces are still subject to many challenges relating to the nature of policy processes as well as to the dynamics of social-ecological systems.

The Complexities of Social-Ecological Systems

Carpenter et al. (2009) admit that synthesizing knowledge about ecosystem services and the way humans interact with and impact upon them, was a daunting task for MA participants. The assessment revealed the enormous complexities of these interactions – or social-ecological systems, as Ostrom (2009) has termed them. Different drivers are influencing social-ecological systems, such as population growth, changing patterns of consumption, and technological innovations. Sometimes these drivers combine to strengthen certain developments, at other times they exert contradictory pressures. Different spatial levels need to be taken into account with some processes having quite localized effects while others cover much larger spatial extents or jump across different spatial scales. Decisions taken in one part of the world, such as, for instance, the growing demand for biofuels in richer countries driven by a need to reduce dependency on fossil fuels and the desire for “green energy”, may affect biodiversity and ecological processes in other parts of the world, resulting, for instance, in the conversion of biodiversity rich tropical forests in South-East Asia into mono-cropped plantations. This may have consequences for certain ecosystem services such as soil retention and fertility, the provision of wild fruits or medicinal herbs, which in turn has impacts on human well-being.

While the example provided above may appear to be a relatively straightforward one, in practice it is rare to find a linear causal path from changes in drivers to biodiversity changes, to changes in ecosystem processes, to impacts on ecosystem services, to consequences for human well-being, to human responses, to feedback to drivers and back to biodiversity again, as Carpenter et al. (2009) illustrate. One of the MA sub-global assessments, the *Caribbean Sea Ecosystem Assessment*, showed how coral reef biodiversity is influenced by indirect drivers (such as urbanization, investment in unsustainable tourism, international shipping practices, fragmentation of authority among 22 island states), direct drivers (land and sea use, coastal pollution, fish harvest, climate change, river discharge, alien species introductions), the demand for certain ecosystem services (principally ecotourism and fish harvest),



and amenity values measured as jobs, GDP, and investment. Part of the complexity of modelling and predicting biodiversity impacts from these influences arises from the different turnover times of crucial ecological and social processes, as well as connections of individual and institutional actions and responses, and ecological changes across all of these multiple dimensions of scale (see also Henle et al. 2010, Liu et al. 2007). The effects of certain forms of coastal pollution may take time to become visible, while other forms may have more direct impacts; tourism may develop rather quickly, while organizing coordination between the various island states to regulate industries or control the impacts of tourism development may take longer. The challenge in this situation is to identify the different processes and turnover times, and strengthen the capacity of ecosystems to support social and economic development (Folke et al. 2004).

The complexities of social-ecological systems render the setting, for example, of biodiversity conservation targets quite difficult. Policy-makers face the challenge to decide whether targets can be achieved through national or regional efforts (Görg et al. 2010), and which targets require more intense collaboration at the international level, including coordination across existing agreements. While it seems obvious that international action is needed with respect to climate change and the management of watersheds straddling international boundaries, it may seem less obvious that also the conservation of certain species may require initiatives at that scale. Henle et al. (2010) cite attempts to protect the common wall lizard, which is rare in Germany and targeted by a number of conservation programmes, while the species is quite abundant in, for instance, Italy. Similar examples are encountered in southern Africa, where some countries, such as Mozambique, are battling to protect elephants, while neighbouring countries such as South Africa are struggling with ever-growing numbers of elephants and have to contemplate culling (Venter et al. 2008). Hence, there is a greater need for coordination of biodiversity conservation targets across the different agreements, and (inter-) governmental agencies, especially as some of the targets may interact. The protection of certain mammals, for instance, also depends on the protection of certain habitats and plant species the mammals feed on, yet creating corridors between protected areas may increase the risk of disease spreading – as it happened when the borders between Kruger National Park in South Africa and Limpopo National Park in Mozambique were opened up and tuberculosis-infected buffaloes from South Africa threatened to infect the tuberculosis-free buffaloes in the Limpopo National Park (Michel et al. 2006). The interactions also concern other international agreements, for example, targets to reduce invasive alien species will require the involvement of the World Trade Organization (WTO) and the parties to the General Agreement on Tariffs and Trade (GATT) (Perrings et al. 2011).

Another challenge to policy-makers is that different ecosystem services require different degrees and forms of diversity. Species that support a service such as climate regulation may be different from species that support, for instance, a service such as food production (Perrings et al. 2011, MA 2005). Certain ecosystems

may also provide bundles of services that are all important for local livelihood strategies – certain forests may contribute to carbon sequestration, groundwater retention and food production at the same time (Bennett et al. 2009). Impacting biodiversity in these cases may enhance one service – for instance, food production – at the expense of other services, hence trade-offs may exist between different ecosystem services. This requires scientists and policy-makers to look at bundles of services and possible trade-offs. Such trade-offs include:

- **temporal trade-offs:** benefits now – costs later,
- **spatial trade-offs:** benefit here – cost there,
- **beneficiary trade-offs:** some win – others lose,
- **service trade-offs:** manage for one service – lose another.

These trade-offs are real and have to be dealt with. The challenge is to look for possible synergies and moving towards “winning more and losing less,” as can be illustrated in relation to the creation of protected areas (box 1).

In terms of beneficiary trade-offs, this entails a need for coordination not only across different environmental agreements and conventions, but also between those and other international agreements such as the Millennium Development Goals (Reid et al. 2010). Box 2 (pp. 130 f.) discusses whether the effects of increased private wildlife conservation indeed contribute to poorer development and biodiversity conservation, as proponents of this strategy argue.

Science-Policy Interfaces and the Politics of Policy-Making

Determining the priorities concerning biodiversity conservation targets are inherently political processes. Science can assist in identifying or predicting what we collectively lose when certain choices are made, though Reid et al. (2010) argue that this requires a much greater effort by science to better serve the needs of decision-makers and citizens, especially in terms of improving the usefulness of forecasting. However, decisions relating, for instance, to the above-mentioned trade-offs reflect national discussions and divergent perceptions of the relative importance of the different ecosystem services and their relative vulnerability (Perrings et al. 2011, Koetz et al. 2009, 2008).¹

Despite the political nature of decision-making relating to biodiversity and ecosystems, the promotion and design of global environmental assessments – such as the MA – is, as already indicated, often based on the technocratic assumption that consensus in science will lead to political consensus, which will then automatically lead to certain policy responses (Koetz et al. 2009, Pielke 2007). Another assumption is that the assessments can serve to develop global norms, which will then cascade down to

¹ The political nature of such decisions came clearly to the fore during the recent CBD COP (Conference of Parties) 10 negotiations (Jessel 2012).

BOX 1:

Evictions and Resettlement for Biodiversity Conservation

The creation of protected areas still results in the eviction or resettlement of local residents (figure 1). Especially in developing countries, where local residents often depend heavily on the natural resources in protected areas, and where land tenure rights are tenuous (figure 2), this results in severe impoverishment (Milgroom and Spierenburg 2008). However, beneficiary trade-offs are possible, if they are taken into account before planning, and attempts are made to study how biodiversity conservation could be combined with fostering local livelihoods and protecting local land rights.

FIGURE 1: Family in Limpopo National Park, Mozambique, waiting to be resettled outside of the park.



FIGURE 2: Sign detailing new restrictions on natural resource use affecting people whose residence and agricultural area was declared a national park without their consultation, Limpopo National Park, Mozambique.



more local levels of government. This model, however, only works when the problem is relatively simple, for instance the problem of ozone depletion (Koetz et al. 2009). Mooney (2002) cites the ozone depletion policies indeed as a successful case of interaction between scientists and policy-makers, which resulted in the *Montreal Protocol on Stratospheric Ozone*, developed in response to new scientific knowledge about the causes of the problem and effects of mitigation strategies. This required flexible policy design, but was facilitated by the fact that a large part of the solution lay in applying new technologies, hence, facilitating technocratic solutions – though economic considerations were also important to some extent. Yet, when realities are more complex, involve more and a variety of trade-offs, and the stakes are higher and decisions urgent, science-policy relations are less straightforward.

There is a long tradition of policy researchers studying how policy-makers deal with complex problems, and very few policy analysts actually subscribe to a technocratic approach to policy-making (Dovers 2005). Especially when there is a sense of urgency, and pressure is exerted upon policy-makers to come up with

solutions, Cohen, March and Olsen (1972) claim policy-makers tend to resort to the “garbage can model”, i. e., the mixing of ends and means in an uncoordinated way in response to short-term imperatives. Etzioni (1967) recognized this as well, but distinguished different phases in the policy-making process; an initial phase of broad scanning of the problem, looking for ways forward, which is then followed by a second stage of more rigorous analysis and policy design. In a seminal article, Lindblom (1959) argued that most policy-making is, almost by necessity, incremental. The complex problems policy-makers are struggling with are difficult to analyse in their entirety, the best anyone can achieve is a partial analysis, arriving at a “bounded reality” (Lindblom 1979). As a result policy changes occur in small steps away from the status quo, and rarely involve radical changes. While Lindblom (1959, 1979) referred to this as “muddling through”, in his 1979 article he clearly stated that this was not meant as a negative normative indication. Experimentation is inevitable and even necessary when dealing with complex problems.

A core problem in analysing policy-making is the tension between policy on the one hand, and politics, political value and the

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state on the other (Dovers 2005). Lindblom (1979) already posited that in all political systems, some degree of “partisan mutual adjustment” takes place, i. e., fragmented or decentralized political decision-making, in which various, more or less autonomous participants mutually influence one another. Policies are affected by a broad range of participants and interests. As a result, the underlying reasons of many policies are obscure, as the various participants in the decision-making process all act according to diverse rationale. Many scientists and stakeholders deride politics, claiming it obstructs rational decision-making, yet, Davis et al. (1993, p. 257) warn that: “Politics is the essential ingredient for producing workable policies, which are more publicly accountable and politically justifiable (...) it is integral to the process of securing defendable outcomes. We are unable to combine values, interests and resources in ways that are not political.”

The relation between politics and global environmental assessments has also been subject to discussion, the question being whether scientific knowledge should be independent or politically legitimated. On the one hand, Koetz et al. (2009, p. 1) argue that the recent “proliferation of global environmental assessments” is an attempt of “putting the onus on the back of science”. Instead of dealing with the politics of environmental policy-making, science is expected by policy-makers to provide an objective, rational, politically neutral body of knowledge (Dickson and Adams 2009). Looking at the role of the scientific advisory board of the *CBD*, the Scientific Body for Scientific, Technical and Technological Advice (SBSTTA), Görg et al. (2010) question this view. They argue that there was a lot of critique on SBSTTA’s role, as this body did not function as an independent scientific unit, but served as a “pre-negotiations table” for the main executive body of the *CBD*, the *Conference of Parties (COP)*. As a result, the SBSTTA was highly politicized in its preparation of documents submitted to the *COP*. On the other hand, there is evidence that results presented by an independent scientific assessment such as the *Geosphere-Biosphere Assessment* mentioned above were rejected by some countries on the grounds that the assessment lacked political legitimacy. This is even true for the *MA*. Although it clearly managed to demonstrate the value of biodiversity, although it was commissioned by

BOX 2:**Synergies or Trade-offs: Is Private Wildlife**

One of the most important biodiversity conservation strategies is the expansion of protected areas. Increasingly, privately-owned lands are incorporated into this strategy. In South Africa this is quite a prominent trend; an estimated 13 to 17 percent of the country’s surface is under some form of private wildlife production. Proponents of game farming present it as a win-win strategy which contributes positively to biodiversity conservation by increasing wildlife habitat, and to economic development by generating foreign currency and providing employment for residents in South Africa’s poverty-stricken rural areas.

A dominant idea is that the possibility of economically exploiting wildlife will increase its value, and hence more landowners will set aside land for wildlife. Economic exploitation takes on various forms: eco-tourism, (trophy) hunting (figure 3) and venison production. However, within South Africa the alleged positive impacts are subject to debate.

Government departments at both national and provincial level have displayed ambivalent attitudes towards game farming. Some officials endorse the win-win vision. Others criticize game farming for wasting productive land to create playgrounds for the rich. The Department of Agriculture (DoA) holds a more ambiguous position with some officials deeming private wildlife production a profitable form of land use, while others fear it will negatively affect food production. The state has benefited from rising revenues, and the Department of Environmental Affairs (DEA) has tended to facilitate the growth of the wildlife industry. An increasing number of species can be hunted in South Africa, the Eastern Cape topping the list with 56. The Eastern Cape province’s permit fees alone amounted to 66,83 million rand in 2009, a figure which does not include other environmental policy revenues such as transportation permits, daily fees or taxidermy exportation fees. The DEA has criticized game farmers though for harmful conservation practices, including intensive breeding of trophy animals and overstocking of land with species that are attractive to tourists. Attempts to introduce stricter environmental policies are contested by landowners who complain it interferes with business opportunities. Politically and legally, the policy shifted wildlife species into new property regimes that facilitate the privatization and commodification of wildlife, thereby contributing to the expansion of game farms. Given the somewhat different mandates and constituencies of different departments, the views on wildlife ranching within the state both converge around some aspects and diverge around others, sending contradictory messages to the various other stakeholder groups who are affected.

In terms of benefits to local economic development, the contribution to employment generation turns out to be exaggerated. In fact, in many cases a process of land consolidation is taking place – a general trend in commercial farming areas in South Africa due to increased competition on the world market resulting from deregulation of the agricultural sector – with more land in fewer hands. On farms where hunting is the main source of income – but also on those where venison production and breeding for live sales is taking place – employment opportunities actually decrease. Many farm workers who not only work but also live on farms lose their jobs and homes. Those who remain are often no longer allowed to keep livestock or cultivate crops – an essential strategy to supplement low incomes – on the game farms. Large-scale private wildlife reserves offering game viewing to the high-end of the tourism market seem to be an exception, these do generate more employment opportunities compared to previous forms of land use; though former farm dwellers are usually confined to the lower echelons. However, these enterprises form a minority of all private wildlife production enterprises, and given the global economic crisis there seems little room for expanding this particular sub-sector of the wildlife industry (Brooks et al. 2011).

FIGURE 3: Hunting trophies at a private game ranch in South Africa. The views on private wildlife conservation are ambiguous.

the United Nations and several biodiversity-related conventions, since it was not strictly organized in an intergovernmental manner, governments did not feel ownership of the results. Hence, the recently launched *Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services (IPBES)* is planned in an intergovernmental way (Larigauderie and Mooney 2010). To increase the political relevance of the *IPBES*, the majority of the delegates discussing the organization of the initiatives in Nairobi and Busan, agreed that it should address urgent societal questions and

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processes. To increase scientific credibility, they called for the best available data, transparency of methods, peer reviewing results, and most importantly that “the assessments need to be unbiased from political pressure” (Görg et al. 2010, p. 184). Emphasis was placed on the need for translating results into policy options. However, in response to some delegations which expressed the fear that a strong scientific panel would interfere with national sovereignty, options should be policy relevant, but not policy prescriptive (Görg et al. 2010).

provide better guidance for policy implementation, enforcement and evaluation.” The importance of an explicit framing of assumptions is confirmed by Huitema and Turnhout (2009). Taking the example of the Dutch Environmental Planning Agency, they show that in assessing various environmental policy options proposed by political parties in the Netherlands, the agency used cost-efficiency as the most important criterion, which actually meant that the assessments were quite prescriptive in terms of demanding an alignment with government’s financial policy without

Pielke (2007) has warned that when scientists refrain from being prescriptive, this leaves decision-makers who are dependent on government agencies, corporations or other interest groups, to (mis-)interpret scientific findings regarding their policy implications. Science can offer a powerful way of framing environmental problems, but it can be mobilized as a source of authority by different parties (Hajer 1995, Ozawa 1996). While this may be true, the framing of environmental problems by scientists is not necessarily neutral either, as the discussion of the role of the SBSTTA or IPCC shows. Moreover, given the complexity of the linkages between biodiversity and ecosystem (services), scientific consensus may also be quite hard to arrive at – though this is still the aim of *IPBES*.

Huitema and Turnhout (2009) show that in practice those participating in science-policy interface platforms, which they refer to as boundary organizations, have to walk a fine line between brokering, advocacy, and being prescriptive. Many struggle to reconcile usefulness with scientific validity, and battle to maintain validity when interaction with policy is inevitable. In general, two strategies are deployed: transparency (in how uncertainties and values are dealt with) and independence. For some, political independence means that they should be able to put issues on the agenda they consider important, and be critical of policies. Others consider that as coming uncomfortably close to advocacy. Koetz et al. (2009, p. 2) also emphasize the need for transparency, and call for “the honest brokering of policy alternatives that systematically presents a broad range of policy options under detailed framing assumptions that pro-

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questioning the economic model underlying that policy – and without making explicit that this model shaped their assessment.

Taking the anxieties of policy-makers about a possible loss of sovereignty seriously as well as scientists' fears about the latter "taking science and running with it," the suggestions made by Koetz et al. (2009) and Huitema and Turnhout (2009) certainly make sense. Part of being explicit about underlying framings should certainly include providing detailed information about the way the various policy options take into account trade-offs – temporal, spatial, beneficiary, and service-related. Furthermore, this equally entails presenting a clear picture of the distribution of costs, risks and benefits among the various stakeholder groups. The suggestions made by some *TEEB* project members (Ring et al. 2010) to develop ways of valuing ecosystem services that focus specifically on poor environmental resource users seems to fit with this perspective as well.

ed in 1983, aims to do so, and was followed by the foundation of a European network, *A Long-Term Biodiversity, Ecosystem and Awareness Research Network (ALTERNET)*, in 2004. The *CBD* has strengthened its connections with scientists, notably with the *DIVERSITAS* programme, an international, non-governmental umbrella programme for research projects, which increasingly has been striving for an integration of natural and social sciences. A relatively new initiative is the *Programme on Ecosystem Change and Society (PECS)*, a research initiative of the International Council for Science and UNESCO. *PECS* specifically aims to address the question how policies and practices affect the resilience of the portfolio of ecosystem services that support human well-being and allow for adaptation to a changing environment. A great diversity of institutional arrangements, policies and practices has been proposed to achieve these goals, yet the difficulty is to evaluate these options (Carpenter et al. 2012). *PECS* will build partly on

Biodiversity scientists need to acknowledge and engage with the political aspects of biodiversity policy-making. They need a better understanding of how institutions to govern ecosystem (services) emerge and are established.

Strengthening Science-Policy Interfaces by Studying Governance

Conducting assessments on a regular basis, as is the intention of the *IPBES*, may provide possibilities for a more flexible, adaptive approach to policy development in response to the successive assessments. However, the translation of their results into concrete policies depends on the political and institutional contexts in the various member states and on power relations between these (Voß and Bornemann 2011).

The *MA* already demonstrated the importance of multi-scale analysis, to study how decisions taken at one level influence biodiversity and ecosystem services at another level. Given these cross-scale interactions, it is, moreover, important to study how decisions are taken at various levels, in other words, the policy-process and institutional arrangements pertaining to biodiversity and ecosystems are of interest. This entails a need for more integrative approaches to biodiversity research including social sciences and so-called informal knowledge. Since the *MA* a number of projects have started along those lines, while ongoing initiatives in that direction are receiving more attention, which could feed into the *IPBES* assessments. One such initiative that involves experimenting with different approaches to data gathering and modelling is the *IHOPE* project, *Developing an Integrated History and Future of People on Earth* (Hibbard et al. 2010). To develop a better understanding of nature-society interactions, long-term place-based social-ecological research is needed, while paying attention to the linkages between the different spatial scales (Carpenter et al. 2009). The *Long-Term Ecological Research Network*, start-

ed in 1983, aims to do so, and was followed by the foundation of a European network, *A Long-Term Biodiversity, Ecosystem and Awareness Research Network (ALTERNET)*, in 2004. The *CBD* has strengthened its connections with scientists, notably with the *DIVERSITAS* programme, an international, non-governmental umbrella programme for research projects, which increasingly has been striving for an integration of natural and social sciences. A relatively new initiative is the *Programme on Ecosystem Change and Society (PECS)*, a research initiative of the International Council for Science and UNESCO. *PECS* specifically aims to address the question how policies and practices affect the resilience of the portfolio of ecosystem services that support human well-being and allow for adaptation to a changing environment. A great diversity of institutional arrangements, policies and practices has been proposed to achieve these goals, yet the difficulty is to evaluate these options (Carpenter et al. 2012). *PECS* will build partly on

ongoing research at certain sites, strengthening inter- and trans-disciplinary research, but will also include new sites and integrated research teams. Research will focus on understanding social-ecological dynamics and transformations, cross-scale interactions, and the management of trade-offs. This focus is a response to calls for more flexible and adaptive, bottom-up policy models (Carpenter et al. 2009, Liu et al. 2007).

An example of this turn to "reflexive governance" is adaptive management, focusing on collective experimentation and learning (Voß and Bornemann 2011). It evolves from the analysis of socio-ecological systems, and seeks to experiment with and improve environmental governance. The aim is to develop, through a bottom-up approach, new institutional arrangements to govern ecosystems (Berkes and Folke 1998). These need to be continuously adapted in response to the effects of the arrangements on ecosystem functioning. The turn to reflexive governance involves shifting perspectives and acknowledging the difficulty of establishing policy goals, inherent uncertainties about effects of alternative options, the agency of the various stakeholders and the power relations amongst them, as well as seeking to integrate a diversity of perspectives, knowledge, expectations, and strategies from different stakeholders. It also has implications for the role of scientists in policy-making (Voß and Bornemann 2011).

However, all reflexive governance approaches unavoidably face a dilemma. On the one hand, there is a requirement to nurture bottom-up spontaneous developments that are open to ambivalence and contestation, and to retain adaptability towards the complex dynamics of change. On the other hand, there remains a requirement to achieve coordination, to take into account a broader

view of developments and long-term goals. Therefore, most approaches to reflexive governance in the end pragmatically combine top-down and bottom-up elements into procedural designs for social learning and policy-making.

Here again, however, it is important not to lose sight of politics. While reflexive governance approaches acknowledge power relations among different stakeholders, they often fail to address other important political aspects of policy-making. These include interactions with the wider political context, the potential of the approaches to marginalize particular interests and social groups, linkages with institutions of representative democracy, and the tendency of adaptive management approaches to stabilize and support an incumbent (capitalist) political economy. In response to this critique Voß and Bornemann (2011) propose to devise specific rules of procedure that enable participating actors to explain diverging understandings and conceptualizations of the problem at hand and reflect on their views in relation to a diversity of others. This may not result in a unified strategy for dealing with social-ecological change, but draws attention to a diversity of perspectives and related interests that are relevant and need to be studied and accommodated.

Concluding Remarks

Policy-makers dealing with environmental issues are confronted with the difficult task of weighing different trade-offs and their different valuations by different sections of their constituency, and trying to predict the outcomes of their choices. The challenge for scientists is to assist in that by studying the implications of such policies for biodiversity, ecosystem services and human well-being. This requires careful study of the dynamics of social-ecological systems, and the different (possible) trade-offs between different ecosystem services, as well as the temporal, spatial and social trade-offs.

Yet, perhaps an even more important challenge for scientists is the need to acknowledge and engage with the political aspects of biodiversity policies (Voß and Bornemann 2011). This requires a delicate balancing act. Policy-makers may feel their mandate is being infringed upon when scientists are perceived to be prescriptive. Scientists, on the other hand, may fear a loss of authority and control over their findings when policy-makers are seen to interpret these to suit their own objectives. Various studies of policy-making processes and science-policy interfaces suggest that the most fruitful way to deal with these tensions is for both scientists and policy-makers to acknowledge the diversity of valuations and preferences among various stakeholders – often related to the same trade-offs scientists can map – as well as power relations between the various stakeholders. Related to this is the proposal of the studies' authors to do justice to this diversity and the principle of transparency through the presentation of a broad range of policy options, and being explicit about, for instance, the economic model and principles that underpin the different options (Koetz et al. 2009, Huitema and Turnhout 2009, Pielke 2007). Such

transparency in turn requires a better understanding of how the various trade-offs are weighed in policy-making, as well as how institutions to govern ecosystem (services) emerge and are established (Carpenter et al. 2012).

The increased acknowledgement by biodiversity scientists of the importance of governance to the study of biodiversity, ecosystems and ecosystem services is currently translated in a number of (relatively) new research initiatives such as *IHOPE*, *ALTERNET* and *PECS*. This in turn reinforces the realization that a much deeper integration of the natural sciences, social sciences and the humanities is needed (Young et al. 2008). This need was confirmed by scientists, policy-makers and practitioners alike during the recent *Planet Under Pressure* conference. Held in London in March this year, it was attended by more than 3000 scientists from all disciplines interested in biodiversity.

References

- Barkmann, J., R. Marggraf. 2010. Zahlungsbereitschaftsbefragungen für Umweltgüter: Wirklich "Finger weg!?" *GAIA* 19/2: 103–109.
- Bennett, E. M., G. D. Peterson, L. J. Gordon. 2009. Understanding relationships among multiple ecosystem services. *Ecology Letters* 12/12: 1394–1404.
- Berkes, F., C. Folke. 1998. *Linking social and ecological systems: Management practices and social mechanisms for building resilience*. Cambridge, UK: Cambridge University Press.
- Bhattacharya, D. K. et al. 2005. Cultural services. In: *Ecosystems and human well-being: Policy Responses*. Edited by K. Chopra, R. Leemans, P. Kumar, H. Simons. Washington, D. C.: Island. 401–422.
- Boyd, J., S. Banzhaf. 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics* 63/2–3: 616–626.
- Brooks, S. et al. 2011. Creating a commodified wilderness: Tourism, private game farming, and "third nature" landscapes in KwaZulu-Natal. *Tijdschrift voor Sociale en Economische Geografie* 102/3: 260–274.
- Brown, T. C. 1987. Production and cost of scenic beauty: Examples for a ponderosa pine forest. *Forest Science* 33/2: 394–410.
- Carpenter, S. R. et al. 2009. Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. *Proceedings of the National Academy of Sciences of the United States of America (PNAS)* 106: 1305–1312.
- Carpenter, S. R. et al. 2012. Program on ecosystem change and society: An international research strategy for integrated social-ecological systems. *Current Opinion in Environmental Sustainability* 4/1: 134–138.
- Cohen, M. D., J. G. March, J. P. Olsen. 1972. A garbage can of model organization choice. *Administrative Science Quarterly* 17/1: 1–25.
- Daily, G. C. (Ed.). 1997. *Nature's services: Societal dependence on natural ecosystems*. Washington, D. C.: Island.
- Daily, G. C. et al. 2000. Value of nature and the nature of value. *Science* 289: 395–396.
- Davis, G., J. Wanna, J. Warhurst, P. Weller. 1993. *Public policy in Australia*. Sydney: Allen & Unwin.
- De Groot, R. et al. 2010. Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation. In: *The economics of ecosystems and biodiversity: Ecological and economic foundations*. Edited by P. Kumar. London: Earthscan.
- Dickson, P., W. M. Adams. 2009. Science and uncertainty in South Africa's elephant culling debate. *Environment and Planning C: Government and Policy* 27/1: 110–123.
- Dietz, T., P. C. Stern. 1998. Science, values, and biodiversity. *BioScience Policy Forum* 48/6: 441–444.

- Dovers, S. 2005. *Environment and sustainability policy: Creation, implementation and evaluation*. Sydney: Federation.
- Duraiappah, A. et al. 2005. Consequences of responses on human well-being and poverty. In: *Ecosystems and human well-being: Policy Responses*. Edited by K. Chopra, R. Leemans, P. Kumar, H. Simons. Washington, D. C.: Island. 487–526.
- Ehrlich, P. R., A. H. Ehrlich. 1981. *Extinction: The causes and consequences of the disappearance of species*. New York: Random House.
- Etzioni, A. 1967. Mixed-scanning: A “third” approach to decision-making. *Administration Review* 27/5: 385–392.
- Feld, C. K. et al. 2010. Indicators for biodiversity and ecosystem services: Towards an improved framework for ecosystems assessment. *Biodiversity Conservation* 9: 2895–2919.
- Folke, C. et al. 2004. Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of Ecology, Evolution, and Systematics* 35: 557–581.
- Görg, C., C. Nesshöfer, A. Paulsch. 2010. A new link between biodiversity science and policy. *GAIA* 19/3: 183–186.
- Hajer, M. 1995. *The politics of environmental discourse: Ecological modernization and the policy process*. Oxford, UK: Oxford University Press.
- Heink, U., I. Kowarik. 2010. What criteria should be used to select biodiversity indicators? *Biodiversity Conservation* 19: 3769–3797.
- Henle, K. et al. 2010. Securing the conservation of biodiversity across administrative levels and spatial, temporal, and ecological scales: Research needs and approaches of the SCALES project. *GAIA* 19/3: 187–193.
- Hibbard, K. et al. 2010. *Developing an Integrated History and Future of People on Earth (IHOPE): Research plan*. Stockholm: IGBP (International Geosphere-Biosphere Programme). IGBP Report 59. www.igbp.net/download/18.1b8ae20512db692f2a680006394/report_59-IHOPE.pdf (accessed May 17, 2012).
- Huitema, D., E. Turnhout. 2009. Working at the science-policy interface: A discursive analysis of boundary work at the Netherlands Environmental Assessment Agency. *Environmental Politics* 18/4: 576–594.
- Jessel, B. 2012. Zwischen Anspruch und Wirklichkeit. Das Übereinkommen über die biologische Vielfalt und sein Einfluss auf die Naturschutzpolitik. *GAIA* 21/1: 22–27.
- Klie, A. 2010. Die Bewertung von Umweltgütern mittels Zahlungsbereitschaft. Woran Kosten-Nutzen-Analysen scheitern. *GAIA* 19/2: 103–109.
- Koetz, T., P. Bridgewater, C. Miller, R. Norgaard, R. A. Pielke. 2009. *Science-policy interfaces for more effective governance of biodiversity and ecosystem services: Institutional mismatches, shifting paradigms, obstructions, and opportunities*. Concept note for IPBES. http://sciencepolicy.colorado.edu/admin/publication_files/resource-2783-2009.40.pdf (accessed May 17, 2012).
- Koetz, T., P. Bridgewater, S. van den Hove, B. Siebenhüner. 2008. The role of the Subsidiary Body of Scientific, Technical and Technological Advice to the *Convention on Biological Diversity* as science-policy interface. *Environmental Science & Policy* 11: 505–516.
- Larigauderie, A., H. A. Mooney. 2010. The *Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services*: Moving a step closer to an IPCC-like mechanism for biodiversity. *Current Opinion in Environmental Sustainability* 2: 9–14.
- Lindblom, C. B. 1959. The science of “muddling through”. *Public Administration Review* 19/2: 79–88.
- Lindblom, C. B. 1979. Still muddling, not yet through. *Public Administration Review* 39/6: 517–521.
- Liu, J. et al. 2007. Complexity of coupled human and natural systems. *Science* 317: 1513–1516.
- Loreau, M. et al. 2001. Biodiversity and ecosystem functioning: Current knowledge and future challenges. *Science* 294: 804–808.
- MA (Millennium Ecosystem Assessment). 2003. *Ecosystems and human well-being: A framework for assessment*. Washington, D. C.: Island.
- MA. 2005. *Ecosystems and human well-being: Synthesis*. Washington, D. C.: Island.
- Michel, A. L. et al. 2006. Wildlife tuberculosis in South African conservation areas: Implications and challenges. *Veterinary Microbiology* 112/2–4: 91–100.
- Milgroom, J., M. Spierenburg. 2008. Induced volition: Resettlement from the Limpopo National Park, Mozambique. *Journal of Contemporary African Studies* 26/4: 435–448.
- Mooney, H. A. 2002. The debate on the role of biodiversity in ecosystem functioning. In: *Biodiversity and ecosystem functioning: Synthesis and perspectives*. Edited by M. Loreau, S. Naeem, P. Inchausti. Oxford, UK: Oxford University Press. 12–19.
- Mooney, H. A., P. R. Ehrlich. 1997. Ecosystem services: A fragmentary history. In: *Nature's services: Societal dependence on natural ecosystems*. Edited by G. C. Daily. Washington, D. C.: Island. 11–19.
- Naeem, S., M. Loreau, P. Inchausti. 2002. Biodiversity and ecosystem functioning: The emergence of a synthetic framework. In: *Biodiversity and ecosystem functioning: Synthesis and perspectives*. Edited by M. Loreau, S. Naeem, P. Inchausti. Oxford, UK: Oxford University Press. 3–11.
- Ostrom, E. 2009. A general framework for analyzing sustainable social ecological systems. *Science* 325: 419–422.
- Ozawa, C. P. 1996. Science in environmental conflicts. *Sociological Perspectives* 39: 219–230.
- Perrings, C. et al. 2011. Ecosystem services for 2020. *Science* 330: 323–324.
- Pielke, R. A. 2007. *The honest broker: Making sense of science in policy and politics*. Cambridge, UK: Cambridge University Press.
- Reid, W. V. et al. 2010. Earth system science for global sustainability: Grand challenges. *Science* 330: 916–917.
- Ring, I. et al. 2010. Challenges in framing the economics of ecosystems and biodiversity: The TEEB Initiative. *Current Opinion on Environmental Sustainability* 2: 15–26.
- Schaich, H., C. Bieling, T. Plieniger. 2010. Linking ecosystem services with cultural landscape research. *GAIA* 19/4: 269–277.
- TEEB (The Economics of Ecosystems and Biodiversity). 2010. *The Economics of Ecosystems and Biodiversity: Mainstreaming the economics of nature: A synthesis of the approach, conclusions and recommendations of TEEB*. Malta: Progress. www.teebweb.org/Portals/25/TEEB%20Synthesis/TEEB_SynthReport_09_2010_online.pdf (accessed May 17, 2012).
- Venter, F. et al. 2008. The evolution of conservation management philosophy: Science, environmental change and social adjustments in Kruger National Park. *Ecosystems* 11: 173–192.
- Voß, J., B. Bornemann. 2011. The politics of reflexive governance: Challenges for designing adaptive management and transition management. *Ecology and Society* 16/2: 9.
- Young, O. R., L. A. King, H. Schroeder (Eds.). 2008. *Institutions and environmental change: Principle findings, applications and research*. Cambridge, MA: MIT Press.

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