Tools for Better Decision Making: Bridges from Science to Policy

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Introduction – Decision making in the context of climate change

There is a priority need in Europe to maintain and improve the quality of our aquatic ecosystems and water resources and the benefits that they provide. Such benefits as clean water, flood risk reduction, fisheries and biodiversity underpin sustainable development, but their links to the natural environment have rarely been core to governmental decision making. The need to reverse this deficiency was recognized in the Plan of Implementation for the World Summit on Sustainable Development at Johannesburg, which has been endorsed by the European Union (EU).

There are three key areas where support is needed: developing strategies for the protection of ecosystems; improving water resource management and the scientific understanding of water; and the promotion of integrated water resource development.

The intricacy of aquatic ecosystems and their connections with land and atmospheric processes make catchment management difficult. Decision making is also complicated because under the EU Water Framework Directive, catchment management must cater for many stakeholders. The need to take account of ecosystem services (e.g. Defra 2007; Royal Society 2009) and implement the 'Ecosystem Approach' (*sensu* Convention on Biological Diversity) makes for further complexity. Uncertainty is inherent in all long-term decision making, but in the case of catchment management and water resources, uncertainty is acutely highlighted by the prospect of climate change.

Uncertainty implies that the outcome of a given policy or management decision cannot be predicted with reliability. Rational decision making handles uncertainty by identifying all possible outcomes and then weighting the value of each with the statistical probability that this outcome will materialize. Thus the choice with the highest expected benefit to one or more groups of people represents the optimal course of action, although this applies only in the immediate future and only takes into account human aspirations. Many or all of the choices included could conceivably be disastrous in a longer term.

This approach, using risk analysis, presumes that we know the probability distribution of different outcomes, which we may not (Kahneman & Tversky 1979). If the mechanisms leading to different outcomes are not sufficiently understood, no one can determine the probability that a particular outcome will follow from a decision (Walker *et al*. 2003). Hence, the poorer the understanding, the more uncertain are the predictions. This presents major challenges to governments and regulatory agencies.

However, uncertainty varies not only by level but also in nature. Uncertainty may reflect inadequate knowledge, or it may reflect inherent variability in human and natural systems (Walker *et al*. 2003). At least three sources of variability may be at play: randomness of natural processes; human behaviour, which often deviates from the rational model of decision making; and interacting social, economic and cultural phenomena. Uncertainty resulting from inadequate knowledge may be remedied or reduced through research. On the other hand, uncertainty owing to inherent variability is beyond management control and cannot be reduced. Uncertainty generally increases as the period of the policy or management decision extends into the future (Brewer 2007). This is because the availability of reliable data diminishes as decisions reach further into the future, and potential variability also increases over longer periods.

For freshwater ecosystems, the understanding of structure and function under current conditions is quite far advanced, and the remaining uncertainty can be reduced through further research. Likewise, the impact of key direct drivers of aquatic ecosystem change is well understood. Such drivers include, for example, temperature, hydrology, nutrients, acid deposition and toxic substances. By comparison, the impact of indirect drivers, such as the effects of climate change on agricultural practices and land use, and other social and economic changes, are less well understood.

Individual and social behaviours vary enormously. For example, the contribution to global food price increases of the recent expansion of biofuels production was not widely foreseen. Land-use patterns and nutrient levels may be affected by policies directly seeking to regulate them, but they are also affected by socio-economic factors that determine the relative costs and benefits of different farming options. Further, human behaviours acting as drivers of ecosystem change are shaped by individual, professional and cultural norms, for instance those concerning good agricultural practices (Nielsen 2009). Thus the impact of human action on aquatic ecosystems represents a significant source of uncertainty.

Climate change is currently the most evident source of uncertainty. Recent assessments of climate changes in Europe conclude that temperatures are likely to increase by 2.1 °C–4.4 °C by 2080 and that precipitation will either increase or decrease depending on the particular region (EEA 2007). Furthermore, it is predicted that extreme weather incidents will become more frequent. However, predictions for both temperature and precipitation changes are characterized as ‘highly uncertain’ (EEA 2007: 152). This uncertainty interacts with the inherent variability of freshwater ecosystems as well human impacts on the drivers discussed above. For instance, climate change may prompt migration of species and new crop patterns, changing habitats, migration of pest species and establishment of alien species. Likewise, extreme weather events might change the resilience of natural systems, for instance to nutrient influxes. Overall, extreme events could lead to non-linear pressures on existing systems, which increase variability and impede predictability. These uncertainties bear on assessments of mitigation costs and benefits. If the long-term effects of mitigation measures cannot be predicted, it will not be possible to estimate the costs of attaining a certain level of protection either.

Recognizing uncertainty as inherent in environmental policy formulation, the EU applies the precautionary principle to decision making. The principle was instituted in EU law with the Maastricht Treaty of 1992 (Treaty on European Union (92/C 191/01)) and is written in Article 174. The principle is not explicitly defined in the treaty, but at its heart is the notion that policy action against potential threats to the environment may be justified even without deterministic scientific proof of harm, as expressed in the German word *Vorsorgeprinzip*, which refers to acting with foresight (Andersen 2000; EEA 2001: 13). The Commission states that the precautionary principle may be applied as ‘a risk management strategy’ in some fields, specifically when scientific evidence is insufficient or uncertain and there are reasonable grounds for concern (Commission of the European Communities 2000: 10). Accordingly, measures adopted should be proportionate with a ‘desired level of protection’ as opposed to zero risk and they should rest on an examination of the benefits and costs of action or lack of it. Finally, the Commission calls for continued scientific evidence to re-examine policy measures. Thus, the EC use of the precautionary principle clearly holds that scientific uncertainty cannot justify a lack of action, even while it holds that scientific knowledge is the *sine qua non* on which to base any decision making.

However, in some circumstances, traditional methods for decision making, i.e. those that aim to identify single optimal choices, may not be appropriate and so alternatives must be found. One alternative is to use scenario-based analyses that use models to explore the consequences of different policy or management decisions under different scenarios. Rather than single-best policies, policy analysts look for robust policies that perform satisfactorily across different scenarios (Walker & Marchau 2003; Popper *et al*. 2005). Furthermore, scenario analysis allows the relative importance of different components to be assessed and makes transparent the trade-offs involved in each strategy. This potentially enables more resilient outcomes.

Scenario building presupposes that, given a particular set of drivers, the future can be predicted well enough to examine the outcomes of different policies across the scenarios. Walker and Marchau (2003) question that robust policies can be identified, particularly for climate change, and therefore question the value of scenario approaches. They advocate instead a stepwise approach: ‘Take those actions now that cannot be deferred; prepare to take actions that may later become necessary; monitor changes in the world and take actions when they are needed’ (Walker & Marchau 2003: 3).

Given this uncertainty, the maintenance of ecosystem structure and functioning should be an overarching management objective. The rationale for more holistic thinking in the management of water to achieve this objective is embedded within the Ecosystem Approach. The Ecosystem Approach is a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way and, as such, it forms a methodological framework for implementing projects that is underpinned by 12 principles. The Ecosystem Approach has been endorsed by the Convention on Biological Diversity (CBD) as the primary framework for the implementation of the Convention. It has also been endorsed by the European Commission at the World Summit on Sustainable Development and by the Ramsar Convention, and its principles are highly congruent with elements of the Water Framework Directive (WFD).

The Ecosystem Approach can help to challenge the natural science community to widen its perspective on how detailed science fits into the overarching policy framework (Maltby 1999). Regardless of methodology, however, the common thread is that action is necessary and that action should be based on the best available evidence. This chapter describes tools and decision-making approaches that should help policy makers and catchment managers define more robust strategies, or prudent first steps, in a world where there may be many impacts on freshwaters resulting from climate change.

Tools for decision making and their bases

Modelling

Computer models have a key role to play in supporting policy decision making. They might be used to predict future ecosystem conditions under a changed climate, or the likely outcomes of mitigation measures, and may help in understanding important ecosystem linkages and processes. There are, however, some potential limitations. Difficulties can arise when models are used to predict beyond the range of conditions for which they are calibrated, or if under more extreme external forcings the system passes a threshold whereby the model parameters and the embedded relationships between them no longer correctly represent the system.

In transferring modelling results to decision making, a key challenge lies in incorporating other sources of data, such as economic data, so that robust decisions can be made that consider the full range of implications of any actions for the environment, society and the economy. Models tend to focus on relatively few variables, although model coupling to create a suite of models within a single modelling framework can increase the range of parameters that can be successfully modelled, together with their linkages (e.g. the modelling of the interaction of nitrate and acidification). However, any management decision concerns the system as a whole and is not necessarily restricted to environmental variables. Evaluating the overall range of effects of climate change, or choosing between alternative mitigation options, requires that the relationships of the target system with wider society and the economy are considered and, potentially, trade-offs made between different outcomes (e.g. installing flood defences with the consequent loss of biodiversity, or allowing floodplains to flood and accepting the potential economic losses).

Establishing the system-wide impacts of a policy intervention or some other driver of change can be difficult, requiring an understanding of cause–effect relationships, including those among, and between, ecological, economic and social systems. It also requires these relationships to be quantified. There are varying degrees to which this can be achieved by use of modelling and other techniques, such as collection of monitoring data, space-for-time substitution or expert judgement. While model coupling can capture critical relationships for a limited number of ecosystem variables, modelling of ecosystems and their interactions with social systems and the economy is not yet possible. Thus techniques are required that allow the outputs from models of different system components to be judged in the context of each other. This does not capture the dynamic nature of some of the relationships between system parameters and is probably less accurate than dynamically coupled models, but it can provide valuable insights.

Linking models – a case study

The case study described here illustrates the use of a non-dynamic linkage between the Climate and Land-Use Allocation Model (CLUAM) and the Integrated Catchment Model of Nitrogen (INCA-N) to generate input data for the application of the Decision Support System (DSS) to the Tamar catchment described below. The objective of the Tamar DSS case study is to model the effects of climate change on diffuse pollution of nutrients within the catchment, and to evaluate alternative mitigation measures. While Global Climate Models can provide estimates of changes to temperature and rainfall under different CO_2 emission scenarios, the effect of such climate changes on land-use patterns could be more significant than the direct climate changes themselves on diffuse pollution. The cause–effect relationship between changes in climate and changes in land use is difficult to assess, affected as it is by both the climate requirements of different crops and the global and regional economics of crop prices, which are governed by the effects of climate change and the agricultural response to it elsewhere.

The CLUAM is a linear programming model of agriculture in England and Wales that provides a formal framework within which to examine likely land-use effects of changes in policy, market conditions and changes in climate (see Hossell *et al*. 1995; Parry *et al*. 1996, 1999; Jones & Tranter 2006). CLUAM treats agriculture in England and Wales as a single ‘farm’ consisting of a range of land types with regional variation. The national ‘farm’ can produce nine crop and four main livestock commodities, such as the main arable crops and meat and milk production, using a range of inputs and resources (e.g. fertilizer and land) on the different land types. CLUAM partitions the England and Wales land base into 15 Land Classes, each containing a mix of land cover types, i.e. arable, ley (short-term grass), permanent pasture and rough grazing, based on the Centre for Ecology and Hydrology’s Land Classification System (Bunce *et al*. 1996a, b). Within each Land Class the land under each of the four land cover types is further subdivided into yield categories, reflecting the range of production potential due to precipitation totals and soil type and, for both arable and grassland, nitrogen input levels. The selection of production activities within each Land Class is constrained by:

1. The availability of land of different qualities within each Land Class (including the possibility of converting one type to another, e.g. ploughing permanent pasture to create arable land) and the ability to switch resources between uses;
2. The total volume of production (reflecting consumer demand) and input use required;
3. Policy constraints that restrict the areas of production activities and input use, or impose specific land-use patterns in certain areas to conform to environmental or other objectives (e.g. quotas, limits to input use in designated areas such as Nitrate Vulnerable Zones).

Within these constraints, land use is determined by the CLUAM according to the maximum profit that can be earned from all the possible activities on all the parcels of land. Both the outputs and inputs, for all the scenarios, were measured in terms of mid-1990s (base year) prices in order to allow direct comparison of results for the future time periods in equivalent value terms.

The CLUAM generates the following outputs, at the Land Class, regional and national levels:

1. Changes in livestock numbers and crop and grassland areas;
2. Areas of land under different land types and the area falling out of agriculture;
3. Areas of land transfers (between land cover types and reflecting land improvement);
4. Change in the use of inputs, including fertilizer and chemical use per hectare and in aggregate.

The CLUAM was adapted for use with river catchments by the inclusion and delineation of the Conwy, Kennet, Tamar and Wye catchments within the model itself. Eight separate model runs have been undertaken. Four of these are ‘reference’ runs and four are ‘scenario’ runs. As the object of the modelling exercise is to capture the effect of climate change on land use, the four reference runs represent the future without climate change, but including projections of future social and economic developments derived from the A2 and B2 climate scenarios described in the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES; IPCC 2000). The four scenario runs are based on the A2/B2 socio-economic futures, but also include the Hadley Centre Climate Model (HadCM) climate change forecasts for the same periods. Comparison of the scenario and reference runs yields the marginal effect of climate change alone. Provided alongside the results of these eight model runs are the results of a further run, called REF1990s, which represents broadly the current position, before recent (Fischler) Common Agricultural Policy reforms, which were implemented in 2005.

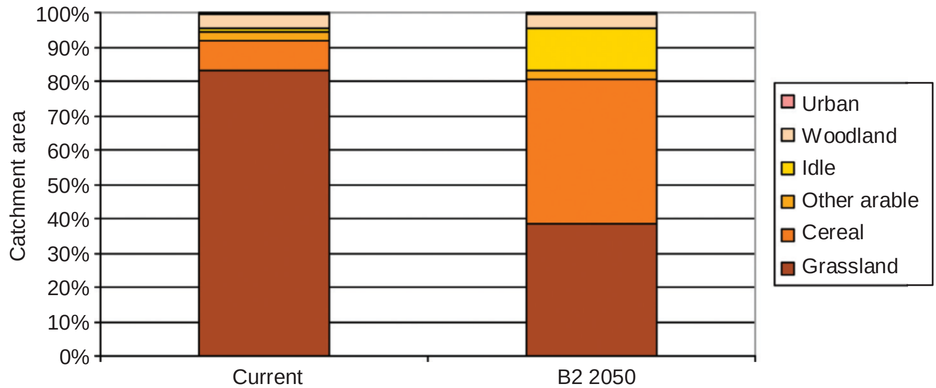

Figure 11.1 Current land-use distribution and projected distribution at 2050 under B2 climate scenario from CLUAM.

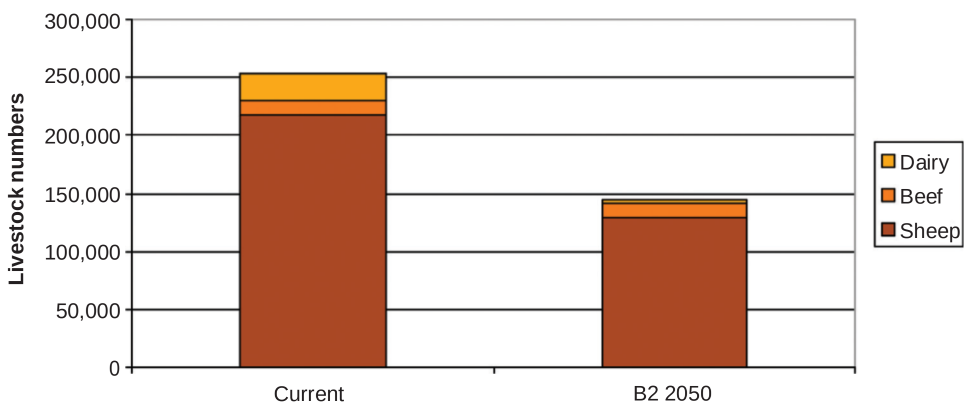

Figure 11.2 Current livestock numbers and projected numbers at 2050 under B2 climate scenario from CLUAM.

In the scenario runs, the CLUAM is driven by the price, demand and technology changes projected by the two global futures (A2 and B2) for both 2020 and 2050, both with and without the HadCM3 climate change projections. However, in the case of the climate change runs, the model is also constrained by the impact of changes to local growing conditions. These local climate changes can have both positive and negative effects, making some areas unsuitable for production of some crops, while making production of 'novel' crops suitable in other areas. For example, lower autumn rainfall levels may make planting of winter cereals difficult in some regions, while a longer and drier summer growing season may make production of crops like grain maize feasible in others.

Figures 11.1 and 11.2 show current land use, as reported in the UK Agricultural census (Defra 2004), and the predicted land use and livestock numbers from the CLUAM in 2050 under the B2 climate scenario. These data provide the inputs to the INCA-N model required to predict the outcome of the direct environmental

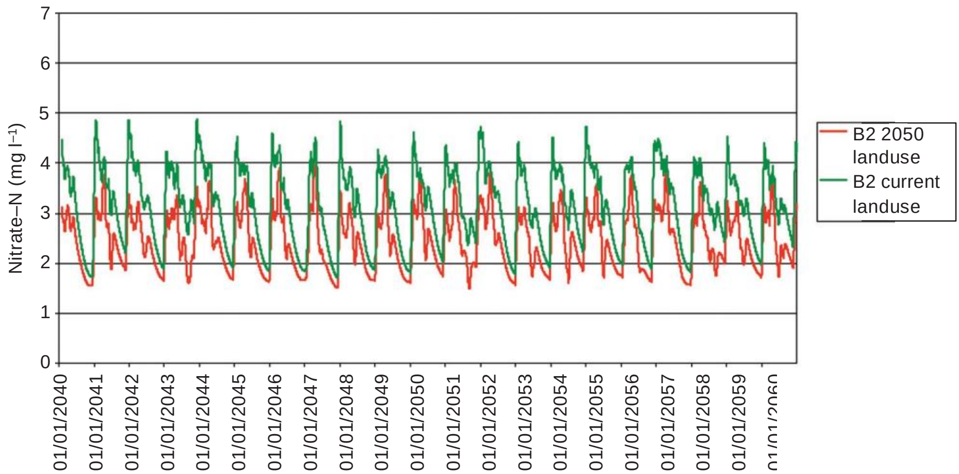

Figure 11.3 Effect of incorporating CLUAM land-use distribution into INCA-N model.

and indirect socio-economic effects of climate change on diffuse pollution. INCA-N, which requires data on organic and inorganic fertilizer application rates, land-use distribution and climate as inputs, predicts *inter alia* daily streamflow concentrations of NO_3–N.

Incorporating the indirect effects of climate change on diffuse pollution in this way can show significant effects on the predicted stream nitrate concentrations from INCA-N. Figure 11.3 shows the stream nitrate concentration at the downstream end of the Tamar catchment (Gunnislake Bridge) for the period 2040–60 under the IPCC B2 climate scenario using the current land use and the land use for 2050 predicted by CLUAM. The CLUAM predicts that the area of grassland and stock numbers in the catchment, particularly of dairy cattle, will be reduced. However, the areas of non-farmed land and cereal land will be increased. Coupled with a predicted reduction in the application of artificial fertilizers of 37% from the SRES projections, the net effect of these changes, as illustrated by Fig. 11.3, is that average stream nitrate concentration will decrease but still remain very high compared with undisturbed conditions.

The use of economic valuation as a tool for decision making

In some regions of the world, water will be an increasingly scarce commodity as precipitation falls and temperature rises. The value of water, however, is not simple to assess, owing to its nature as a 'public good'. Here we explain how the total economic value (TEV) of changes in the quantity and quality of water can be estimated. Such an assessment is integral to the design of economic incentives and institutional arrangements to ensure wise allocation of resources and sustainable management, now and in the future as the impact of climate change increases. The issues involved are illustrated by presenting a case study for the Cheimatitida wetland, located 40 km Southeast of Florina in Northwest Greece, a region where future water resources will be especially threatened by climate change.

Because they are not traded, many of the goods and services generated by lakes, rivers and marginal wetlands are hard to quantify in monetary terms and the consequent risk of their being neglected in policy making has received attention from economists. Water is one of the most important natural resources on which development and indeed survival are based. It is indispensable for domestic use and at the same time of vital importance to agriculture and industry.

Water resources have been degraded and depleted through shifts in climate and by more direct human activities. Integrated water resource management, linking social and economic development with protection of natural ecosystem functioning, is needed if sustainable development is to be achieved. To design and implement efficient and effective policies for water resource management, the TEV of the benefits generated by its several services and functions needs to be determined. Given that many of these benefits (including indirect ones such as amenity, or general ecosystem support) are not reflected in market prices, economists attempt to estimate the true resource value with the use of alternative valuation techniques.

TEV distinguishes between the value that individuals derive from consuming the environmental resource (use values) and the value that they derive even if they do not use it (non-use values). Use values can be classified into direct use, indirect use and option values. Direct use values come from consumption as drinking water, irrigation or industrial raw material. For most private goods, value is almost entirely derived from their direct use. Many environmental resources, however, perform functions that benefit individuals indirectly: indirect use values of aquatic habitats include benefits such as flood control, nutrient retention and storm protection. Finally, option value recognizes that individuals who do not currently use a resource may still value the option of using it in the future. The option value for water resources therefore represents their potential to provide economic benefits to human society in the future.

Non-use values (Krutilla 1967) can be classified into existence value, bequest value and altruistic value. Existence value refers to the value individuals may place upon the conservation of an environmental resource, which will never be directly used by themselves or by future generations. Individuals may value the fact that future generations will have the opportunity to enjoy an environmental resource, in which case they might express a bequest value. Altruistic value means that even if the individuals themselves may not use or intend to use the environmental resource themselves, they may still be concerned that the environmental good in question should be available to others in the current generation.

The challenge for economic valuation is to assign money value to non-market goods and services and thus assist policy makers in determining policy priorities. Over the last several decades economists have developed and refined methods for estimating the non-market values of goods and services. These non-market valuation methods can be categorized as revealed and stated preference methods, depending on whether they are based on existing surrogate markets or constructed hypothetical markets.

Revealed preference methods work by analysing actual markets that are related to the non-market resource under valuation. Information derived from observed behaviour in the surrogate markets is used to estimate willingness to pay (WTP), which represents an individual's valuation of, or the economic benefits derived from, the environmental resource. The essence of the Stated Preference (SP) approach is that the market for the good is 'constructed' through the use of a hypothetical scenario. Consequently, stated preference techniques circumvent the absence of markets for environmental goods and services by presenting consumers with hypothetical markets in which they have the opportunity to pay to use or protect, or accept compensation for the loss of the environmental good or service in question. Two approaches used to derive such values are the Contingent Valuation (CV) method and the Choice Experiment (CE) method.

In a CV study, respondents are asked their maximum WTP (or minimum willingness to accept (WTA) in compensation) for a predetermined increase or decrease in environmental quality. They are offered a change in the quantity or quality of a good at a given cost, and either accept or refuse the payment of the suggested cost. To provide an accurate WTP measure, the survey must meet the accepted standards of survey research (Arrow *et al*. 1993). In a CE study, the environmental resource is defined in terms of its attributes and the levels these attributes would take with and without sustainable management of the resource. Attributes of a water resource could include biodiversity and water quality, whereas levels of biodiversity could, for example, include the number of endangered bird species conserved. A monetary cost/benefit attribute is also included to allow for the estimation of WTP or WTA values. Experimental design methods (Louviere *et al*. 2000) are used to construct different profiles of the environmental good in terms of its attributes and levels of these attributes. Two to three such profiles are assembled in choice sets, which are in turn presented to the respondents, who are asked to state their preferences in each choice option. Statistical analysis is used to estimate the factors that affect choice of an environmental good as a function of its attributes and the levels these attributes take. Respondents' WTP or WTA for a change in any one of the attribute levels is calculated as the trade-off they make between the level of that attribute and the monetary cost/benefit attribute.

The WTP or WTA values estimated from CV and CE studies can then be used in Cost Benefit Analysis (CBA). CBA is an analytical tool based on welfare theory, which is conducted by aggregating the total costs and benefits of a project or policy over both space and time (Hanley & Spash 1995). A project or policy represents a welfare improvement only if the benefits net of costs are positive. Different management options will yield different net benefits and the option with the highest net benefits is the preferred or optimal one.

A CE was implemented to address the issue of efficient management of the Cheimatitida wetland (Birol *et al*. 2006a). Based on expert consultations, literature review and discussions with local people, four wetland attributes that are expected to generate non-use values were selected. These were: (i) biodiversity; (ii) open water surface area; (iii) inherent research and educational values that can be extracted from the wetland; and (iv) values associated with environmentally-friendly employment opportunities. The levels these attributes would take, with and without sustainable management efforts, were determined with wetland experts at the Greek Biotope and Wetland Centre. The fifth attribute was a monetary cost attribute, levels of which were determined through a previous CV study regarding this wetland (Birol *et al*. 2006b). Using these five attributes and their levels, experimental design methods were employed to generate choice sets containing alternative wetland management scenarios and an option to select neither scenario. An example of a choice set is presented below.

Sample choice set

Which of the following wetland management scenarios do you favour? Option A and option B would entail a cost to your household. No payment would be required for 'Neither management scenario' option, but the conditions at the wetland would deteriorate to low levels for biodiversity, open water surface area and research and education attributes, and no locals would be re-trained.

	Wetland management Scenario A	*Wetland management Scenario B*	Neither management scenario A nor management scenario B:
Biodiversity	Low	High	
Open water surface area	Low	Low	
Research and education	High	Low	I prefer NO wetland management
Re-training of locals	50	50	
One-off payment	€3	€10	
I would prefer: (Please tick as appropriate)	Choice A □	Choice B □	Neither □

The CE survey was administered in February and March of 2005 with face-to-face interviews with 407 members of the Greek public located in eight towns and two cities. These locations were selected to represent a continuum of distances from the Cheimaditida wetland, as well as rural and urban populations. The public's WTP for improvements in each one of the four attributes were estimated and it was found that on average the Greek public is willing, on average, to pay €7–8.4 per person for conservation of high levels of biodiversity; €6.5–10 for provision of higher levels of their open water surface area; €3.2–6.2 for investments in education and research; and €0.07–0.17 for re-training of a local farmer in an environmental-friendly employment. These WTP values represent the economic benefits the Greek public derives from higher levels of these attributes, which when combined represent the cost they are willing to pay for sustainable management of the wetland. When these economic benefits were compared with the costs of providing higher levels of these attributes, it was found that the benefits far exceeded the costs, which means investments in sustainable management of this wetland would bring about welfare improvements in Greece.

The results indicate that there are positive and significant benefits to the sustainable management of the Cheimaditida wetland. The impacts of social, economic and attitudinal characteristics of respondents on their valuation of wetland management attributes were also found to be significant, implying that there is considerable difference of opinion within the Greek public, which should be taken into consideration when assessing the provision of public goods, such as wetlands (Birol *et al*. 2006a).

Transfer of science into policy

Policy makers steer the direction of research by means of research funding policy. For example, in the European Research Programmes FP5 and FP6 several projects have been funded to create knowledge and develop methods for the implementation of the WFD. In FP6 stronger emphasis has been laid on global change. This underlines the possibilities for the EU Commission to channel research activities according to policy needs. The same applies in the member states. However, the link between science and policy is very variable among member states and within different aspects of water management.

Scientists may improve the exchange of knowledge by focusing on research topics that are relevant to the needs of society and policy/decision makers (Quevauviller & Thompson 2005), though in effect this may often be development of pre-existing research, i.e. applied research, rather than new fundamental research. There is no *a priori* way that fundamental research can be predicted to be relevant or not. Appeals to provide 'useful' knowledge, i.e. with direct policy implications, are often aggravating for scientists. The principles of academic freedom may lead to research outputs that do not meet the immediate requirements of policy making in a way contract research would do. Yet, even when not engaging in user-orientated research, scientists may contribute to policy development by entering into public discussions about their research to help define policy problems and to influence future funding (e.g. Day *et al*. 2006).

In the field of water policy, three groups of users can be identified: policy makers, decision makers at the operational level and the public at large. Each of these groups has different information needs. Policy makers need current information on drivers and their impacts; they also need information about the expected effects of possible policies, as well as the costs of implementing them. Further down the line, policy makers will need reviews of the responses of humans and ecosystems to policy instruments, to assess their efficacy.

Decision makers, who are responsible for the practical implementation of policy, require more specific and detailed information on methods, technologies and good practices (Quevauviller & Thompson 2005). Practical tools and models that derive from research activity are useful at this level. Regarding the WFD, methods for involving the public in decision making will likely be much sought after by operational managers. A significant challenge in transferring science into policy is the sectoralism frequently encountered among decision makers and a lack of trans-disciplinarity among scientists. This can lead to conflicting policy objectives (for instance, conflicts between the objectives of agricultural support payments and biodiversity conservation) and insufficient attention paid to ecosystem linkages, and linkages between the environment and society. Attempts are being made to address these challenges by implementing an ecosystem approach in both research and policy formulation.

Finally, the general public will need general information about water quality and water quantity problems and how climate change may affect water quality and water quantity. Such information would enable citizens to follow the debate around water policy making and implementation and make them more responsive to policy instruments. Generally, transparent decision making tends to increase legitimacy of the decisions made. This assessment, therefore, points to the need for a diverse set of tools and methods for the communication of scientific knowledge.

The role of Decision Support Systems

Given the challenge of integrating science into decision making, techniques are needed that can help decision makers balance social, economic and environmental objectives. Such tools should include Decision Support Systems (DSS). Recently a number of reviews of the use of DSS in water management have been carried out by, for example, Horlitz (2006), Evers (2008) and Giupponi *et al*. (2007). The latter developed 'Guidelines for the development, implementation and application of DSS tools'. A common conclusion of all three works is that users of DSS tools should be involved in their development from the outset. Ideally, they would be involved in financing development projects as a way of increasing their motivation to both develop useful tools and subsequently use them. Many such development projects have encountered problems through lack of practical expertise in support of the research team. This is one of the reasons for the frequent failure of take-up of DSS tools by water managers and policy makers (Giupponi *et al*. 2007). Steps can be taken (Fig. 11.4) to improve this situation. Ideally, designers and users should meet at the start to discuss the objectives and contact should be maintained throughout. Funding for workshops and other expenses is needed and training of users in use of the DSS should start as early as possible.

In some cases there are unforeseen limits to the levels of cooperation that is possible between designers and intended users. Currently water managers are busy implementing the WFD and are very reluctant to take on the additional work of incorporating other considerations, such as the possible consequences of global change. This is perhaps where scientists have to take the lead and emphasize the extent and implications of the evidence now accumulated.

The Euro-limpacs Decision Support System

As part of the Euro-limpacs project, a DSS has been developed to evaluate catchment management strategies in the context of climate change. The DSS provides a GIS-based framework for integrating social, environmental and economic data through Multi-criteria Analysis (MCA). The DSS is implemented as an extension within the computer program ArcGIS. The resulting framework is intended to address specific and targeted management questions such as:

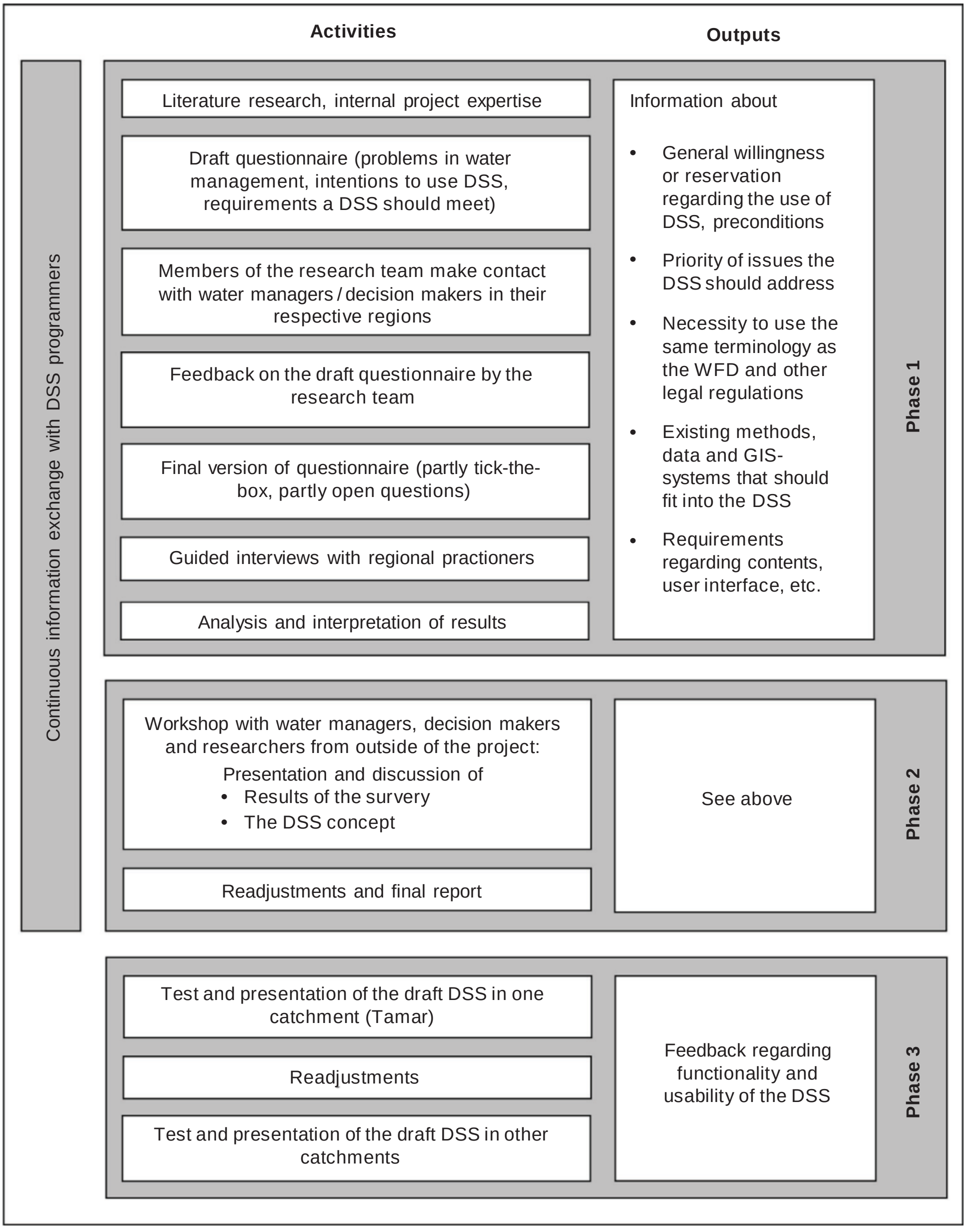

Figure 11.4 Activities to incorporate end-users' requirements into the design of the Euro-limpacs DSS.

- Will climate change affect some parts of a catchment more than others?
- What measures should be taken to mitigate the effects of climate change?
- Which part of a catchment should resources be targeted towards?
- Which measures most effectively tackle the defined problem?

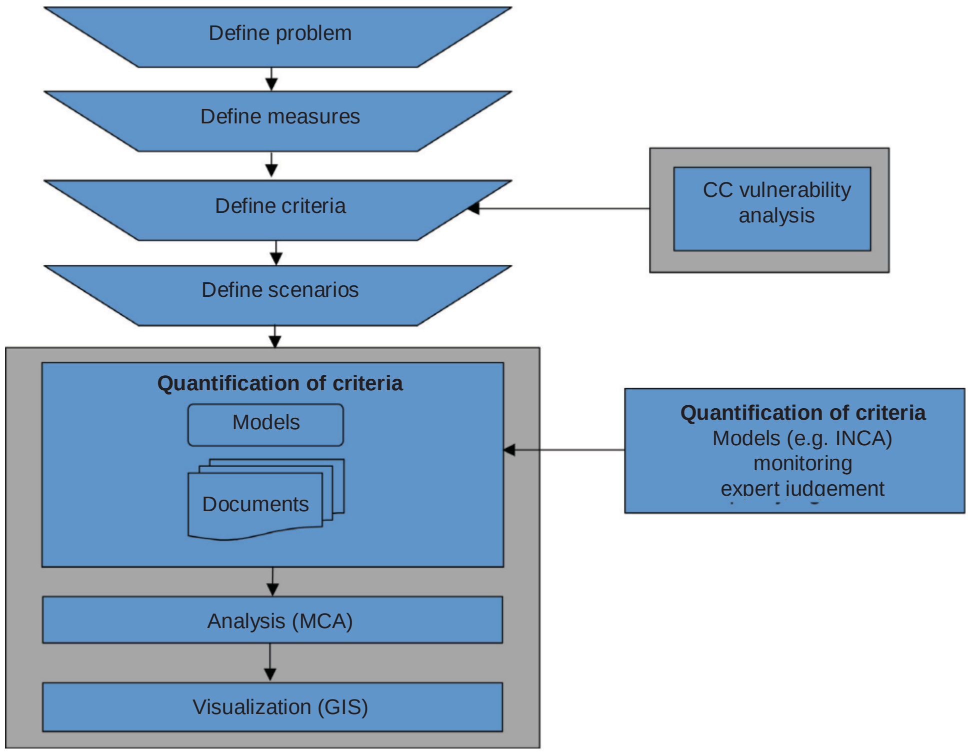

Figure 11.5 Conceptual model of the Euro-limpacs DSS showing the pre-application phases of problem definition and the main phases of the software application. Shaded areas represent stages of the application of the software tool.

There are several steps that the user of the DSS is required to follow, prior to the use of the software tool itself, in order to ensure that the problem to be addressed is structured in a suitable manner (Fig. 11.5). The first step is the definition of the problem in general terms. For instance, it may be defined as diffuse pollution from agriculture, or riparian flooding exacerbated by land-use changes. This step is intended to set the boundaries for the analysis and define the objective of the management strategies to be assessed.

Following this, the user identifies the potential measures that could be put in place to address the problem. Such measures could, for example, be reduction of fertilizer applications, reinstatement of riparian wetlands or reafforestation of upland areas. Summarized reviews of policies relating to different catchment management problems are available within the DSS that illustrate measures previously used.

Criteria are then defined for comparing alternative management options and these include those environmental, social and economic variables that are influenced by the measures that might be put in place. The user selects, from a predefined list, the types of ecosystems that are present in the area in question (e.g. small lakes, large rivers, wetlands) and the DSS tool indicates the ecosystem variables that are likely to be sensitive to climate change. This information has been collated from an extensive review of literature and expert judgement (Hering *et al.* Chapter 5, this volume). The tool also suggests relevant variables that might be used for monitoring these effects and provides a qualitative rationale for the processes involved. In this way the DSS guides the user to include those elements of the catchment that are sensitive to climate change into their analysis.

Using the information from this vulnerability analysis, and current management problems within the catchment, the final preparatory step in the DSS application is the definition of the scenarios, or management measures, that are going to be compared using the DSS. These might be combinations of climate scenarios and management measures applied to different extents. For instance, users may wish to compare the effects of 50%, 100% or 200% increases in the area of riparian wetland under different IPCC climate scenarios.

As the catchment area is divided into subunits (sub-catchments, administrative units, etc.), these preparatory steps in the application of the DSS result in a set of matrices, one for each scenario and management strategy, showing the decision criteria for each spatial unit. The primary purpose of the DSS is to provide a framework to facilitate a structured spatial analysis of these matrices. The user populates the matrices by quantifying the decision criteria for each scenario. This is done outside the DSS and can use a variety of different data sources and approaches, including models, databases or expert judgement. The particular mixture of information used to quantify the decision criteria will vary from application to application, depending on the available data and models.

Once the decision criteria have been quantified, the MCA tools within the DSS can be applied. The value of each decision criterion is converted to a score between 0 and 1 according to a function that is determined by the user of the DSS. The function is intended to convey the relative preference for achieving a certain value for the decision criterion. The form of these functions can be linear or non-linear depending on the decision criteria being considered. Normalizing the scores to the range 0–1 allows decision criteria with different units and relating to environmental, social or economic considerations to be integrated within a single analysis.

The decision criteria are also assigned weights. These weights are set by the user and reflect the importance of the particular decision criteria to the overall comparison of scenarios. For instance, the costs of implementing the measures may be the overriding consideration and this decision criterion would be given a higher weighting than others. Different interest groups may have different priorities and these can be reflected in the choice of weights to determine if the optimal management action is sensitive to the views of different groups. The final MCA score is then calculated as a sum of each of the individual weighted decision criteria scores. The results of the MCA are presented graphically, overlaid on the GIS map showing the spatial units of the assessment.

The DSS has been tested for seven catchments across Europe, of which one is presented here.

Tamar case study

The River Tamar is a small river flowing southwards across the south-western peninsula of England and separating the counties of Devon and Cornwall. Within the catchment, diffuse nitrate pollution from agriculture is a problem and was chosen as the issue to be addressed in this case study application of the DSS.

The management measures selected for representation within the DSS were realistic and reflect previous and ongoing management interventions to address diffuse pollution in the catchment:

- reduction in fertilizer applications and improved fertilizer practices
- reduction in stocking density
- shift from arable to pasture
- restoration of wetlands

Decision criteria were then chosen that could be used to assess the conditions of the sub-catchments under different scenarios of the application of these measures. The decision criteria were:

- nitrate concentration at the outlet of each of the major sub-catchments: yearly average and summer average (as this is an ecologically sensitive period)
- costs of implementing the measures
- biodiversity indicators: area of non-farmed land and area of wetland
- hydrological parameters: mean flow, high flow (Q5) and low flow (Q95)

Three different management options were selected for the application, differing in the extent to which each of the management measures outlined above are applied. These scenarios were:

- Business as Usual (BAU) – Current management and development policies continue unchanged
- Policy Targets (PT) – Policy targets are achieved. This is a 'moderately green' strategy
- Deep Green (DG) – Restoration of the catchment and sustainability are management priority

The climate change scenario years 2050 and 2085 and IPCC climate scenarios A2 and B2 were chosen for use in this exercise. The A2 scenario is characterized by increasing global population, a move towards self-reliance at the national scale and regionally orientated economic development. The B2 scenario is characterized by increasing population, but at a lower level than A2, with a greater emphasis on sustainable development at a local level. When overlaid on the three management options, therefore, there are in total 12 climate/management combinations. For comparison, the current conditions are included as a climate/management combination, making a total of 13.

For each of the climate scenario/management combinations the decision criteria were quantified. These decision criteria were quantified under each scenario/ management combination using modelling and expert judgement. Water quality and water quantity criteria were modelled using the INCA-N model. INCA-N predicts stream nitrate concentration and flow using input data for land-use distribution, fertilizer application rates for different land-use types and daily climate variables (Whitehead *et al*. 1998a, b; Wade *et al*. 2002). Climate data for the A2 and B2 SRES scenarios were provided by Sveriges Meteorologiska och Hydrologiska Institut (SMHI) derived from the ECHAM General Circulation Model (Roeckner *et al*. 2003). The ECHAM model outputs were downscaled to the Tamar catchment using the method of Wade *et al*. (2008).

Data on land use, fertilizer application and stocking density for the BAU management options under the A2 and B2 climate scenarios were taken from the CLUAM outputs and modified according to the scheme set out in Table 11.1 for PT and DG options. Table 11.1 summarizes the data sources and assumptions made to quantify the driving forces for diffuse pollution under each climate/ management combination. These data then formed the input data to the INCA-N model.

The area of non-farmed land is provided as an output from the CLUAM described above. Data for the potential maximum area of floodplain wetland is taken from a survey of wetland areas (Hogan *et al*. 2001) and this figure was used for the DG option. Current wetland areas are estimated to be approximately 25% of historical extent (Hogan *et al*. 2001). These data were modified according to the scheme set out in Table 11.1 for different management options.

The costs of implementing the measures are estimated using the Gross Margin figures for different agricultural activities calculated by the CLUAM. The cost of implementing the measures for BAU options are considered to be zero, as this is the laissez-faire option. Other management options are costed according to the estimated loss of total gross margin resulting from implementing the management measure. For instance, if there is a reduction in head of cattle between the A2 2050 BAU option and the A2 2050 PT option, then the cost is estimated as the reduction in head of cattle multiplied by the gross margin per head of cattle estimated by the CLUAM in 2050. The assumption is that in order to encourage the reduction in stocking density as set out in the option definition table, the cost, in terms of grants or subsidies to landowners, would have to at least equal the loss of income compared with the BAU option for that particular climate scenario.

Figure 11.6 shows results for the Current, Business as Usual, Policy Targets and Deep Green scenarios under IPCC A2 and B2 climate scenarios for 2050. It illustrates the potential usefulness of the DSS to decision makers at the catchment scale as it shows that, under the A2 climate scenario, putting in place progressively more stringent packages of the management measures defined in the scenarios (Business as Usual, Policy Targets and Deep Green) can improve the overall MCA score, indicating a more positive outcome in some of the sub-catchments. Conversely, under the B2 climate scenarios, the total MCA score decreases with progressively more stringent packages of the management measures in most

Table 11.1 Definition of scenarios and management options

	Current	*Business as usual for A2 and B2 scenarios*	*Policy targets for A2 and B2 scenarios*	*Deep green for A2 and B2 scenarios*
Fertilizer applications	Survey of fertilizer practice (Goodlass & Allin 2004)	Predicted by CLUAM	Lower of: 20% reduction on CLUAM prediction or Nitrate Vulnerable Zone limits	50% reduction on CLUAM prediction
Stocking density	Calculated from Defra (2004)	Predicted by CLUAM	Lower of: 15% reduction on CLUAM prediction or Hill Farm Allowance levels	60% of CLUAM prediction
Change landuse from arable to pasture	Current land-use distribution calculated from Defra (2004)	Predicted by CLUAM	50% reduction in arable area predicted by CLUAM	80% reduction in arable area as predicted by CLUAM
Wetlands	No change on current wetland extent.	50% loss on current wetland extent	50% of floodplains restored	All floodplains restored (Hogan *et al*. 2001)
Non-farmed land	Current land-use distribution calculated from Defra (2004)	Predicted by CLUAM	50% increase, or 1% of catchment area, whichever is the larger	100% increase, or 2% of catchment area, whichever is the larger

sub-catchments. This is primarily because there is a much higher proportion of cereals predicted under the B2 scenario compared with the A2 scenario by the CLUAM (0.1% and 42% of the farmed area for A2 and B2 Business as Usual scenarios, respectively), so encouraging the necessary transition to grassland to achieve the management targets in the Policy Targets and Deep Green scenarios is more expensive and reduces the preference for these management options. This suggests that under a B2 climate scenario, measures other than those represented in the DSS scenarios should be employed.

Disaggregating the analysis by sub-catchments allows decision makers to target management measures within the catchment. For instance, presenting the outputs as in Fig. 11.6 highlights sub-catchments where the management intervention is of greatest benefit. Under the A2 climate scenario, for instance, the management measures assessed in the DSS are most effectively applied in the middle reaches of the catchment.

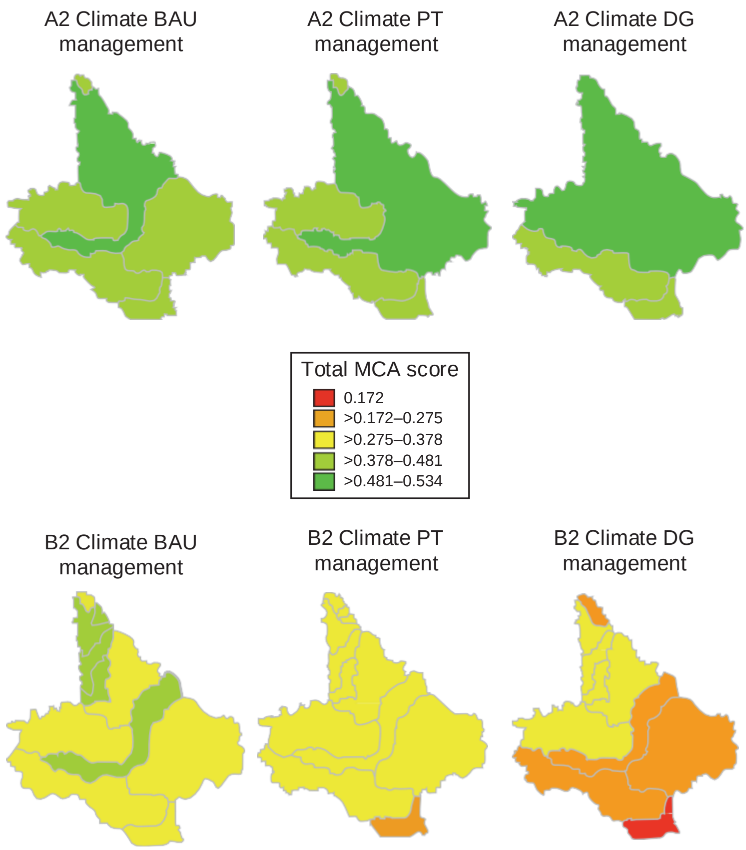

Figure 11.6 DSS outputs for Current, Business as Usual, Policy Targets and Deep Green scenarios under IPCC A2 and B2 climate scenarios at 2050. Maps are coloured according to categorized total MCA scores for each sub-catchment and management/climate combination. A higher score indicates a greater preference of decision makers for that option.

Conclusions

The uncertainty inherent in making policy and management decisions about natural ecosystems presents challenges to both decision makers and to scientists. This uncertainty is set to increase as the consequences of climate change become apparent, making it more imperative than ever that new knowledge gained from research helps to inform policy and decision making. One of the key challenges for scientists is to translate their research into forms that make it useful for decision makers. This will require the development of tools for decision makers, and so these should be high-priority outputs from research projects and programmes. A range of science-based tools that can be applied to managing freshwater ecosystems under climate change already exist, some examples of which are presented here. However, because of their complexity, not all of the available tools are suitable for practical application by decision makers themselves and require application by experts. The chief requirements of decision makers for tools are simplicity, rapidity, transparency and reliability.

A key challenge for decision makers is to act on new scientific evidence and modify current policies and practices accordingly. The potentially rapid shifts in ecosystem conditions that may result from climate change make it imperative that decision makers are responsive to emerging scientific evidence and act quickly. This process would be helped by a greater engagement with the scientific community to guide the development of new tools required for decision making and policy development.

Building bridges between science, policy makers and stakeholders is another important challenge that has to be met if better decision making in the context of climate change is to be achieved. Links between policy makers and stakeholders are becoming more prevalent and embedded within policy development and implementation practice. Such participation has, for instance, been included in the implementation of the WFD and is also a key element of the Ecosystem Approach. New tools to facilitate this are being developed and the Euro-limpacs DSS provides one such example.

Information and approaches from many sources and perspectives should be integrated when addressing current and future freshwater ecosystem management problems. Within the scientific community, integration between the natural sciences and the social sciences, including economics, is essential in order to fully appreciate the indirect social and economic effects on freshwater ecosystems that might result from climate change. These indirect effects can often arise unexpectedly both because of a lack of relevant information on outcomes and because of the uncertainty inherent in human responses to climate change, as embodied in the range of IPCC climate scenarios.

The growing importance of the concept of ecosystem services in national and EU-wide policy frameworks will also make this scientific interdisciplinarity essential: social scientists are needed to understand societal choices and preferences, natural scientists to quantify and understand ecosystem functioning, economists to value the services derived from those functions and the costs of producing and protecting them, and policy makers to develop policy mechanisms to preserve or enhance ecosystem service delivery. All of the tools outlined here, and others, will be needed to achieve this.

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