ECONOMICS FOR ACCOUNTABILITY IN COMMUNITY-BASED ENVIRONMENTAL GOVERNANCE

Working Paper 2 from the project 'Improving economic accountability when using decentralised, collaborative approaches to environmental decisions'

Graham R. Marshall, Judith M. McNeill and Ian J. Reeve

Institute for Rural Futures,
University of New England, Armidale

June 2011

ACKNOWLEDGEMENTS

The research documented in this working paper was funded by the Australian Government's Commonwealth Environmental Research Facilities (CERF) program. The views and judgements expressed in this working paper do not necessarily reflect those of the Australian Government or the CERF program. Feedback on version 1 of this document from David Pannell is gratefully acknowledged. The authors are responsible for any remaining deficiencies in the document.

INFORMATION ON THIS PROJECT

Further information on, and documents from, the project *Improving Economic Accountability when using Decentralised, Collaborative Approaches to Environmental Decisions* is available from http://www.ruralfutures.une.edu.au/staff/3.php?nav=Program%20Leaders&staff=Dr%20Graham%20Marshall

NOTE

This final version of the working paper supersedes version 1 dated May 2009.

CONTENTS

LIST OF TABLES

LIST OF FIGURES

SUMMARY

Background

Australian governments currently devolve significant powers to regional organisations in respect of deciding how public funds available for environmental management should be invested. These regional organisations are part of a community-based approach to environmental governance that governments have justified on the basis of potential benefits for developing the capacities of landholders and other stakeholders to respond self-reliantly (both as individuals and in groups) to the environmental challenges they face. Benefits from this approach have been also expected to flow from its potential advantages in fostering collaboration between stakeholders. Collaboration has been expected to move stakeholders towards increased commonality of value systems and thus towards greater 'community ownership' of decisions that are made.

Calls to strengthen the economic accountability of this collaborative, community-based approach to environmental management have intensified in recent years. The present project – 'Improving economic accountability when using decentralised, collaborative approaches to environmental decisions' – was motivated by concerns that conventional approaches to economic accountability are ill-suited to capturing the kinds of benefits for which the community-based approach was introduced. It was motivated too by evidence that the conventional approaches exceed the capacities of most regional and other community-based environmental organisations to apply.

The purpose of this document was accordingly to lay the foundations for identifying in the present project an approach to economic accountability that is: (a) consistent with stated reasons for adopting a community-based strategy for environmental management; (b) cost-effective to apply given the capacities of community-based organizations; and (c) consistent with an 'economic way of thinking'.

Overview

Three categories of methods for maintaining economic accountability can be distinguished: (i) benefit-cost analysis (BCA); (ii) multi-criteria analysis (MCA); and deliberative methods. These methods are described and reviewed in chapters 2, 3 and 4, respectively. Within each of these categories, we found multiple methods of a more specific nature or purpose.

The development and details of the Investment Framework for Environmental Resources (INFFER) is discussed in chapter 5. This particular framework was singled out for review because it was designed specifically to be applied by community-based environmental organisations, and because it has found significant levels of endorsement and adoption by governments and community-based bodies in Australia. The process of developing this framework also provides key insights into the political economy of designing a method for economic accountability with realistic prospects of accommodating the diverse needs of stakeholders in community-based environmental governance.

Some specific methods for maintaining economic accountability were found to span two or more categories. For instance: deliberative MCA spans the categories of MCA and deliberative methods; deliberative monetary valuation spans the categories of deliberative methods and BCA; and methods have been applied also that incorporate BCA results into an overarching framework of MCA. Although the Investment Framework for Environmental Resources (INFFER) focuses primarily on introducing a 'BCA mindset' into the process of evaluating environmental investment decisions, it also provides

considerable scope for deliberation by communities and other stakeholders in defining problems and solutions and valuing the benefits of the solutions that are identified. Scope also exists to incorporate INFFER output on the relative economic efficiency of projects into a broader MCA framework capable of weighing up a wider range of criteria (e.g., equity, sustainability).

Institutional economics of choosing accountability procedures

Methods of evaluating decisions – including those about environmental investment – are 'value articulating institutions'. Such institutions define the rules to be followed in the process of evaluation (e.g., concerning rights to participate, and how data is obtained and handled).

Choosing a method to use in evaluating a particular environmental investment decision is thus a choice between alternative institutions. Economists have long concerned themselves with institutional choices in general, and the consensus now in neoclassical welfare economics is that such choices are evaluated most appropriately using a comparative institutions approach in which the relevant choice is between institutional arrangements that would be feasible in the relevant context. It is mistaken, therefore, to expect any single method for economic accountability to be optimal everywhere that environmental investment decisions are undertaken.

In respect of economic valuation, this mistake has been made most commonly by advocates of BCA, who tend to compare alternative evaluation methods not with BCA as it would *feasibly* be practised but with BCA as it would *ideally* be practised. A comparative institutions approach recognises that all evaluation methods are imperfect and the objective is to identify the best one given the situation at hand.

Comparing methods for economic accountability

The broad-ranging discussion in this document highlights the wide range of criteria that commentators on evaluation methods for environmental decisions have stated or implied should be considered when choosing between such methods. A non-exhaustive (and non-mutually-exclusive) list of such criteria is presented in the following table. The comments in the table suggest that none of the BCA, MCA or deliberative types of evaluation method is likely to be scored highest across all the criteria, even by someone particularly committed to one of the method types. Hence, a choice between the methods would normally depend on the relative importance that a chooser places on each of the criteria and on how highly they rank or score each method type against each of the criteria.

The focus in the present project is on identifying a method for economic evaluation of environmental investment decisions that is consistent with the stated premises on which Australian governments have devolved significant powers over such investment decisions to regional and other community-based organisations. A method for economic evaluation of environmental investment decisions needs to be consistent with these premises, and be capable of accounting for these kinds of benefits, if it is not to work against the prospects of realising the stated aspirations of Australian governments. This is the standard of objectivity that leading economic thinkers have set for economists; namely, that economists take the goals set by a client (e.g., government, community-based body, etc.) as given and, with minimum subjective input, advise the client how best to achieve those goals.

A selection of criteria for choosing between methods for evaluating environmental decisions	
Criterion	**Comments**
Consistency with neoclassical welfare economics	An advantage claimed for BCA. However, MCA methods based on multi-attribute utility theory (MAUT) share some of the theoretical foundations of BCA. Some stakeholders may view consistency with neoclassical welfare economics as a disadvantage.
Alignment with an economic way of thinking	An advantage claimed for BCA, although MCA can be structured along the lines of an economic way of thinking.
Consistency with a focus on environmental outcomes	An advantage sometimes claimed for BCA, although each of the methods can be applied consistently with a focus on environmental outcomes. The possible advantage for BCA in this respect derives from its use of (shadow) market prices to value decision impacts, where these prices are reasoned to reflect the marginal social utility of any outcome. MCA and deliberative methods are not as constrained theoretically to focus on outcomes, so a risk exists that they may stray from this focus. This risk may be managed with structured procedures that serve to maintain an outcomes focus.
Ability to account for incommensurable values	A weakness of BCA. MAUT-based MCA methods score different values against relevant scales, but subsequently combine the scores into an overall score. Non-MAUT-based MCA methods do not combine scores into an overall score. Deliberative methods are well-suited to accounting for incommensurate values.
Scope to reflect philosophy of integrated environmental (catchment) management	In principle, each of the methods has equal scope to account for the whole range of benefits that a project may generate for different natural and environmental resources (and also in respect of other social and economic assets). In practice, however, BCA likely to face greater problems in accounting for those benefits that cannot be readily valued using existing market prices.
Acceptability of underlying value judgements (to governments & other investors, and to communities & others whose cooperation is required)	Such value judgments may relate to (a) commensurability of different values; (b) appropriateness of measuring environmental and other intangible values on a monetary basis; (c) procedural fairness and distributive fairness (equity).
Acceptability of results (to governments & other investors, and communities & others whose cooperation is required)	Perceived validity of BCA results can be lessened by controversy concerning any non-market values incorporated in the analysis. On the other hand, governments and other stakeholders may prefer the clear-cut answers from BCA to the conditional answers often emerging from MCA or deliberative methods. (However, stakeholders can also be sceptical of clear-cut answers to problems they know are far from simple). Acceptability of results to a stakeholder may also depend on the degree to which they have participated in the process of deriving them, and thus gained trust in and ownership of the results. To the extent that deliberative and MCA methods typically allow for greater stakeholder participation than BCA, their results may find greater acceptance.
Internal consistency	An advantage claimed for BCA because it applies a single coherent body of theory in identifying and measuring all relevant values. Internal consistency in applying a method reduces risks of double counting or of introducing biases in measuring different values. However, structured approaches to MCA can strengthen internal consistency in its application.

A selection of criteria for choosing between methods for evaluating environmental decisions (continued)

Criterion	Comments
Protection against strategic manipulation	An advantage claimed for BCA due to (a) the rigour imposed by its reliance on a single coherent body of theory, (b) its adherence to the principle of individual sovereignty which limits inclusion of values to those of defined (sets of) individuals, and (c) its provision of clear-cut answers which provide discipline on decision makers. Even so, considerable scope for strategic manipulation remains due to reliance of BCA analysts on scientists and others for the data and assumptions they require, and also due to the 'black box' nature of BCA techniques.
Transparency	An advantage claimed for MCA (sometimes referred to as a 'glass-box' method), especially where simpler MCA algorithms are applied. BCA techniques are sometimes criticised for their 'black-box' nature, and deliberative methods have been criticised for lacking transparency in terms of clarifying all the value judgements and reasoning upon which a decision was made.
Ability to account for transaction (including political) costs	BCA often has difficulty here. MCA offers scope to include transaction costs of options as a distinct criterion.
Ability to accommodate values arising from community-based or other deliberative processes	BCA's adherence to the principle of individual sovereignty renders it unable to accommodate values arising from deliberation. In contrast, deliberatively-determined values can readily be facilitated and accounted for by deliberative methods or deliberative approaches to MCA.
Affordability	Financial costs in applying BCA can be high when consultants need to be paid, and particularly when sophisticated non-market valuation (NMV) techniques are employed. Costs of NMV may be reduced by using benefits-transfer methods where appropriate data is available, although validity concerns may arise. MCA and deliberative methods can involve significant financial costs when consultants are required and/or sophisticated techniques for applying these methods are employed. Reliance on deliberative methods (including within MCA) can also be costly in terms of time demands on participants.
Ease of use	Considerable skill is required in applying each of the types of methods – BCA, MCA and deliberative – to a standard at which confidence in the results is justified. Structured procedures may be developed that facilitate ease of application by non-experts. The skill level needed for BCA studies involving sophisticated NMV techniques is higher again, and is not easily alleviated by providing structured procedures.
Timeliness	Limited availability of appropriately-skilled practitioners can lessen the timeliness with which any of the methods is applied (e.g., in respect of funding-submission deadlines). A need to use sophisticated NMV techniques when applying BCA may especially cause timeliness issues. Reliance on deliberative methods (including within MCA) may lessen timeliness when problems arise in coordinating involvement of the relevant participants.
Facilitation of stakeholder learning	An advantage claimed for deliberative methods and MCA methods (particularly where deliberative processes are incorporated). This advantage arises from the greater participation of decision makers and other interested parties in the application of these methods, compared with the more expert-driven process involved in applying BCA.
Ability to reduce conflict and facilitate cooperation	An advantage claimed particularly for deliberative methods, but also for MCA methods that incorporate deliberative processes.

A selection of criteria for choosing between methods for evaluating environmental decisions (continued)	
Criterion	**Comments**
Consistency with recognition that the decisions at issue are concerned with complex adaptive systems	BCA recognises only mechanistic relationships. Unless it forfeits its internal consistency, therefore, it cannot account for consequences of decisions arising from complex adaptive systems (e.g., related to resilience, adaptive capacity, path dependence, irreversibility). MCA and deliberative methods are not similarly constrained.
Scope to account for effects of decisions on ongoing social capacities	Social capacities (e.g., trust, reciprocity, social norms, peer pressure) typically emerge from interactions within complex adaptive systems. Unless BCA forfeits its internal consistency, therefore (see above), it is unable to account for the consequences of a decision for the social capacities needed for longer-term success in environmental management.

An economist committed to providing objective advice on what evaluation method is most consistent with stated aspirations for collaborative community-based environmental management would consider a broader range of criteria than would a mainstream environmental economist. This broader set of criteria may include all 19 criteria identified in the table, and possibly additional criteria relevant to specific contexts. This economist would need to weight the criteria 'alignment with an economic way of thinking' more highly than 'consistency with neoclassical welfare economics', and be sensitive to the expectations of her client/s when weighing up the relative importance of each of the other criteria. With a minimum of subjective input, she would need also to elicit from her clients their judgements of how each evaluation method would perform against each of the criteria.

Towards pluralism in economic evaluation

The scope for such an objective process to choose between economic evaluation methods depends on how 'economic' is understood by the client who desires evaluation of this kind. If the client understands this term as synonymous with the logic of neoclassical welfare economics, or views BCA as the only valid method of economic evaluation, then the scope of the process is constrained to choosing between variants of the BCA method (e.g., between different techniques for non-market valuation). To ensure that a client's choice of BCA is well-informed, the limitations of this method in accounting for the benefits of collaborative community-based environmental management would need to be explained up-front to the client.

Many politicians, policy makers and community leaders do continue to regard BCA (together with cost-effectiveness analysis) as the only legitimate approach to economic evaluation. Some of these are not aware that alternatives to BCA exist, and that growing numbers of economists are promoting and applying these alternatives. Such economists have come to accept the need for a more pluralistic approach to economic evaluation. Moreover, there is a steady increase in recognition among governments and other stakeholders that methods of evaluation other than BCA can be applied consistently with an economic way of thinking, and that these methods can sometimes be more appropriate.

The Investment Framework for Environmental Resources (INFFER) is a notable recent development in these respects. This framework adheres to an economic way of thinking without imposing on decision makers the kinds of value judgments that underpin conventional applications of BCA and cost-effectiveness analysis. It borrows from MCA the idea of a scoring-based approach to valuing benefits

from investments, and leaves community-based environmental organisations ample scope to employ deliberative methods when value judgements are required from them. Despite its accommodation of a more pluralistic approach to economic evaluation, this framework is finding acceptance among government agencies in Australia that traditionally have regarded BCA as synonymous with economic evaluation. Even so, INFFER could be modified in a number of ways (as identified in chapter five) to enhance its suitability for community-based economic evaluation of investment decisions.

1. INTRODUCTION

1.1 Pressures for performance assessment in Australian community-based natural resource management

Expenditure by Australian governments on natural resource management (NRM) programs has grown steeply in the last three decades. The $A340 million allocated in 1990 for a decade of expenditure under to the National Landcare Program represented a quantum leap in the level of public funding of NRM (Hajkowicz et al. 2006b). In their 1989 joint submission to the Commonwealth recommending this program, the Australian Conservation Foundation and the National Farmers Federation stated their aim was to ensure 'Australia's agricultural and pastoral lands are used within their capability by the year 2000 and that there is sustainable use of lands from that time on' (quoted in Toyne et al. 2000 p. 5).

Even so, the Australian Government found itself in 1997 allocating a further $A1.25 billion over five years to NRM programs under the Natural Heritage Trust (NHT). In 2000, the Council of Australian Governments endorsed the National Action Plan for Salinity and Water Quality (NAP), which involved the Australian and state/territory governments committing another $A1.4 billion over seven years to salinity- and water-quality-related programs. In 2001, the Australian Government extended the NHT for a further five years (becoming known then as NHT2, as distinct from NHT1 for the first phase) by contributing an additional $A1 billion (Marshall 2008c). Most recently, in 2008, the Australian Government committed to investing $2.25 billion over the first five years of its new NRM program Caring for our Country (CfoC) (Australian Government 2008).

Compared with an $A11 million spent on the National Soil Conservation Program in 1988-89 (McDonald et al. 1993), budgeted expenditure for 2008-09 under CfoC is 40 times larger (in nominal terms) at $A440 million (Hajkowicz 2009). Australia, unlike most other OECD nations, does not have large amounts of public funds allocated to agricultural production support programs that could be diverted into agriculture-oriented NRM (or 'agri-environmental') programs. Hence, increasing expenditure on NRM programs in Australia has been more challenging politically since it has relied more on reallocating funds from other areas of public expenditure (ibid.). Given this challenge, it is hardly surprising that pressures to demonstrate 'value for money' from expenditures on NRM programs have increased with the magnitude of these expenditures.

One consequence of such pressures has been calls for improved performance assessment of the nation's main NRM programs. The Industry Commission (1998) proposed that accountability under NHT1 be enhanced by establishing goals and performance indicators for each decision-making level (Commonwealth, state/territory, regional and local) against which performance measures could be measured. In its review of the regional delivery model by which the NAP and the NHT2 had been rolled out, the Australian National Audit Office (ANAO) (2008 p. 22) noted that its audits 'in 1996-97, 2000-01 and again in 2004-05 found weaknesses in the monitoring and reporting of the performance of the NHT. In summary there was no effective outcomes reporting'. Moreover, it found overall from its current review that:

> … the information reported in the DAFF [Department of Agriculture, Fisheries and Forestry] and NHT Annual Reports has been insufficient to make an informed judgement as to the progress of the programs towards either outcomes or intermediate outcomes. There is little evidence as yet that the programs are adequately achieving the anticipated national outcomes … (ibid. p. 16).

1.2 Increasing emphasis on priority setting

Another consequence of these pressures has been increasing emphasis on 'priority setting' in deciding how much of the public funds available at any time should be allocated to NRM programs, and how the funds allocated to NRM programs should be directed between competing opportunities. Priority setting has been defined as 'the task of selecting a subset of issues, policies or projects towards which limited resources will be directed. … The priority-setting task always involves trade-offs due to political, social, cultural, financial, legal and technological constraints' (Hajkowicz et al. 2006b p. 88). Crean (2003) argued that NRM should compete with other government portfolios in the prioritisation of public funds, and Edwards and Byron (2001) reasoned that budgets for environmental conservation should be subjected to the same level of scrutiny that applies to budgets for other areas of public expenditure.

The language of priority setting features strongly in policy documentation for the regional delivery model for NRM, under which most of the investments and expenditures have been undertaken by community-based regional organisations. For instance, the Intergovernmental Agreement for the NAP (Australian Government 2001 p. 3) required that an integrated regional plan be developed for each NRM region designated as a priority under the NAP, and that each such plan 'be based upon a scientific analysis of natural resource conditions and problems and priorities carried out at the catchment / regional level …'. Notwithstanding this requirement, the ANAO (2008 p. 24) review of the regional delivery model found that 'stronger targeting … towards the highest priorities and most critical national assets is necessary to achieve measurable results'. In apparent acknowledgement of this criticism, the new CfoC program was announced claiming that '[f]or the first time [the Australian Government] will work towards one clear goal with clearly defined outcomes and investment priorities' (Commonwealth of Australia 2008b p. 22).

1.3 Calls to strengthen economic accountability

This growing emphasis on priority setting has been accompanied by increasing use of language influenced by an economic way of thinking. For instance, the Intergovernmental Agreement (IGA) for the NAP (Australian Government 2001 p. 4) specified that 'catchment / regional targets will be … based on good science and economics' and that these targets will be '… achievable in a cost effective way' . It prescribed also that '[i]nvestment principles for determining priority funding of regional activities should include … the cost-effectiveness and return on investment measured against catchment / regional targets …' (ibid. p. 8). One of the IGA's principles was identified as 'restor[ing] degraded landscapes where that is practical and economic' (ibid. p. 2). More recently, publicity materials for the CfoC program observed that '[h]istorically there was a raft of programs for managing natural resources that … have been unable to demonstrate value for investment'. The new program would 'take a business approach to investment' under which the Australian Government will 'choose the most efficient and effective ways of taking action …' (Australian Government 2008).

The influence of economic thinking is evident also in state / territory level policy documentation concerned with regional NRM delivery. For instance, the New South Wales (NSW) Natural Resources Commission prepared a Standard for Quality Natural Resource Management against which regional bodies in that state (known as Catchment Management Authorities or CMAs) will periodically be audited (Natural Resources Commission 2005). The Standard's purpose 'is to give confidence to the public, government, other interested parties and to natural resource managers themselves that investment in natural resource management is cost effective … and maximises benefits …' (ibid. p. 2).

One outcome required by the Standard concerns monitoring and evaluation, and guidance for this outcome includes the statement that '[e]valuation should assess the efficiency, effectiveness and appropriateness of strategies in progressing towards catchment and state-wide targets …' (ibid. p. 13). In addition, the NSW Government (2006 p. 121) set the following target under Priority E4 (concerned with natural resource management) of its State Plan: 'Natural resource decisions contribute to improving or maintaining economic sustainability and social well-being'.

Despite the increasing emphasis of Australian governments on priority setting within regional community-based processes of investing public funds, and the apparent calls for priority setting to be guided, at least in part, by an economic way of thinking, these governments have been silent on how priority setting of this kind would feasibly occur. Farquharson et al. (2007 p. 3) observed accordingly that:

> …the rhetoric of Governments in Australia for NRM is of 'maximising returns', 'maximising the efficiency and effectiveness of investments in natural resources', and 'targeting resources to the activities and places with the greatest potential for improvement'. However, the processes for achieving these 'goals' are not clearly specified or determined. The CMAs are aware of the rhetoric but do not have guidelines on what constitutes 'maximum efficiency', 'better NRM' or 'maximum return'.

However, this lack of government guidelines seems not to be the only important constraint on regional NRM organisations pursuing an economics-influenced approach to priority setting in their investment planning processes. Marsh et al. (2008) reported that a low level of capacity to assess and apply economics was found in a study of 18 regional NRM organisations around Australia. They reported further that the minority of such organisations using economics 'did so to a modest extent and in a very limited way, mainly through the conduct of benefit-cost analysis of specific programs or interventions' (ibid. p. 12). They concluded accordingly that '[i]t would be valuable, where possible, to involve economists and social scientists in their committee structures. Governments should consider ways to support this' (ibid. p. 14). They reported that 'a number of the interviewees were aware of the need for more economic and social information to inform decision-making' (ibid. p. 12). This is consistent with the scoping phase of the present project finding that staff in the three CMAs studied were aware of the lack of economic rigour in their investment planning processes, and willing to consider using an economics-influenced framework or process developed in this project that is feasible for them to apply, adds transparency and rigour to their decisions, and strengthens their accountability to investors (Marshall 2008b).

1.4 The present project

This document is an output of the project 'Improving economic accountability when using decentralised, collaborative approaches to environmental decisions'. This project was designed recognising the increasing demands for economic accountability in deciding priorities under the regional delivery model, as well as the continuing difficulties that governments, regional NRM bodies and economists have been experiencing in developing and applying methods capable of satisfying these demands.

Three main reasons for these difficulties were identified. The first is that the method of benefit-cost analysis (BCA) conventionally employed for economic evaluation of environmental decisions exceeds the capacities of most regional bodies to apply. Some of the most important effects of environmental decisions are generally not priced by market transactions, and conventional economic methods tend to

rely on estimating these 'non-market values' through sophisticated procedures requiring considerable input from skilled professionals. Given the substantial number of priority-setting decisions normally made by any regional body in any year, it seems unrealistic to expect any such body to apply conventional methods to more than a small subset of these decisions.

A second main reason for these difficulties is that the regional delivery model was intended as a community-based approach to environmental governance, under which considerable emphasis would be placed on developing the capacities of regional bodies and their communities to respond self-reliantly to the environmental challenges they face. Important among these capacities is the social capital shared by groups and individuals. The *National Natural Resource Management Capacity Building Framework* endorsed by the Natural Resource Management Ministerial Council (2002 p. 1) for the regional delivery model stated that 'in addition to the transfer of technology and technical capability, capacity building should foster social cohesion within communities, and build both human and social capital'. It recognised that social capital in the context of regional NRM delivery had multiple aspects including trust, reciprocity, social norms and community ownership. However, the positive-feedback (or increasing-return) dynamics by which social capital is created or destroyed cannot be explained or predicted by conventional economic methods which do not account for these dynamics but only for negative-feedback (or diminishing-return) dynamics (Marshall 2005)[1]. This 'blindspot' of conventional economic analysis renders it deficient in accounting for the full range of implications that governments have identified as important under the regional delivery model.

The third main reason for these difficulties is that decision-making under community-based governance is meant to be collaborative. A key reason for collaboration is its potential to move different parties deliberatively towards collective agreement on the knowledge and value systems they can use in defining their shared problems and deciding how to prioritise alternative solutions. A key assumption of community-based governance is that 'community ownership' of decisions reached under such governance will arise to the extent that community members feel they have been genuinely engaged in a collaborative decision-making process that has accommodated adequately their own knowledge and values. The *National Natural Resource Management Capacity Building Framework* states accordingly that a 'strong feeling of ownership over the NRM planning process will increase motivation and the likelihood that the outcomes identified in the regional integrated NRM plans are achieved' (Natural Resource Management Ministerial Council 2002 pp. 5-6).

The critical difficulty here in applying conventional methods of economic evaluation to community-based environmental governance is therefore that these methods are unable to accommodate collective value systems emerging from a deliberative process of collaboration. This is because these conventional methods remain faithful to classical liberalism's commitment to protecting individuals' rights of privacy (Ezrahi 1990). Conventional economic methods remain faithful to this commitment via adherence to the principle of individual (or consumer) sovereignty. This principle holds that the preferences of individuals should not be influenced by others but rather be formed independently (Norton et al. 1998).

---

[1] Diminishing-return (or negative-feedback) dynamics dampen the effects of an original change such that a system is predictably brought back into equilibrium. Increasing-return (or positive-feedback) dynamics amplify the effects of an original change (e.g., loss of trust) such that the system may not converge to equilibrium or, if it does, it becomes virtually impossible to predict which equilibrium state will occur (Arthur 1999). Economists also refer to increasing-return dynamics as 'cumulative causation', which 'describes a relationship between an initial change in an independent variable and a dependent variable, whereby the dependent variable in turn causes a change in the formerly independent variable in the same direction as the initial movement' (Schmid 2004 p. 112).

Nevertheless, conventional economic analyses of policy decisions do require a collective value system (often called a 'social welfare function') to compare the various options against. The 'ordinalist revolution' within conventional (i.e., neoclassical) microeconomics during the 1930s left this economic tradition without a way of objectively deriving such a collective value system from the value (or preference) systems of individuals. This impasse was circumvented through the technical manoeuvre of adopting 'Pareto efficiency' (commonly referred to as 'economic efficiency') as the conventional economic yardstick of social welfare. A change to the status quo is warranted by this yardstick only when the change represents a 'Pareto improvement' – i.e., it makes at least one person better off without making anyone worse off.

This criterion was found too onerous in the real world where virtually all policy options make at least a few individuals worse off. Hence, it was superseded for practical purposes by the '*potential* Pareto improvement' criterion. According to this more lenient criterion, a proposed change should be regarded as increasing economic efficiency if it makes some people better off and if these people could *potentially* compensate all losers from the change and still remain better off than they were before the change.

1.5 The relationship between government and policy economists

It is perhaps necessary to explain why the second and third reasons provided above for the difficulties faced in demonstrating economic accountability under the regional delivery model are reasons that warrant attention in the present project. In other words, why should economists concern themselves with the expressed objectives of governments when they tend to be wary of 'government failure' and thus more committed to what they conventionally regard as the 'higher calling' of economic efficiency? A first reason is that the justification for pursuing economic efficiency as a higher priority than pursuit of government objectives is flawed. This justification is based on the assertion that economic efficiency provides an objective basis for comparing the social welfare implications of pursuing alternative objectives and the corresponding observation that governments choose their objectives subjectively to an appreciable degree (e.g., on the basis of political and bureaucratic considerations).

This justification is flawed because, as explained above, economic efficiency is not an objective measure of social welfare since it relies on various subjective judgements of value. One of these is that the marginal utility of any given amount of money (e.g., ten dollars) is the same across all individuals irrespective of how rich or poor they might be. Hence, the incidence of benefits and costs of a change is assumed to have no effect on social welfare (Bromley 1989).

The justification is flawed also because, as highlighted by Bromley (ibid.), the notion of objectivity sought conventionally in economics misunderstands what objectivity actually requires. As he explained: 'It is *not* the science − nor the conclusions − that are objective but rather the economist who stands between theory and the individual(s) who must make a decision with economic content and implications' (ibid. p. 233, original emphasis). This standard of objectivity takes the objectives chosen by a client (e.g., government or community-based organisation) as given and requires economists to advise the client how to achieve these objectives with minimum subjective input from themselves. Different economists following this standard should therefore arrive at similar advice for the client.

A second reason for the economics discipline to concern itself with the expressed objectives of governments when evaluating alternative options is that governments are more likely to turn to other disciplines for advice than yield to conventional economic arguments that policy options should be

evaluated against the yardstick of economic efficiency rather than against the requirements of stated government objectives. In a presidential address to the American Agricultural Economics Association urging economists to engage with the sustainable development issue that governments worldwide had embraced, Sandra Batie (1989) argued accordingly that 'policy economics should not be independent of the political expression of society' and warned policy economists accordingly against '[c]lose minded adherence to our ideological convictions … [I]f we cling too tightly to conventional neoclassical concepts we are in danger of trivializing important global problems' (ibid. pp. 1097, 1098-99).

This is not to argue that economists should refrain from scepticism regarding the motives of governments and their capacities for rationally identifying their objectives. Rather, it is to urge greater recognition with the economics discipline and profession that, in most of the world, democratically-elected governments are entrusted with the responsibility of setting the objectives of public policy and ensuring these objectives are pursued competently. As Schmid (1989 pp. 286, 306) argued:

> There is no dichotomy between being efficient and being political. … [T]here is no way politicians can regard BCA as independent information to be weighed, somehow, along with other inputs to make a decision. BCA is either the politician's decision, or it is nothing at all.

Hence:

> BCA [benefit-cost analysis] is not a device for telling government what it must do to avoid being labelled irrational (often stated as being 'political'). … It is not a way to prevent market (or governmental) failure. Failure is relative to someone's goals, and when goals conflict, calling something a failure is only to choose sides. BCA can illuminate the character of the conflict and make it easier for observers to tell whether, once the conflict is settled, the resolution is in fact truly implemented. … Explicitness and consistency are the main values incorporated into BCA (ibid. pp. 285-286).

Samuels (1989 p. xvii) reasoned similarly that BCA:

> … is no substitute for politics understood as self-government and as a mode of working out collective decisions. Indeed, the great genius of BCA is not to determine compulsive, unique optimal solutions to problems but to facilitate the coherent identification and juxtaposition of competing subjectivities and their respective implications.

1.6 The economic way of thinking

The project 'Improving economic accountability when using decentralised, collaborative approaches to environmental decisions' acknowledges the foregoing three main difficulties of applying conventional methods of economic evaluation to priority setting under a regional delivery model which aspires to be community-based. It seeks to explore beyond conventional methods to develop an approach to priority setting that is not only (i) consistent with stated government objectives in adopting community-based approaches for this realm (e.g., advantages for building social capital and for fostering collaborative decision-making), and (ii) feasible and cost-effective to apply given the capacities of community-based organisations, but also (iii) consistent with the 'economic way of thinking'.

The fundamentals of the economic way of thinking, at least as applied to priority setting, relate to the textbook definition of the economic problem as the problem of allocating scarce means among

competing ends. Central to these fundamentals are the concepts of 'opportunity cost' and 'marginal analysis' (Black 2008; Mooney et al. 1997). Since means (e.g., natural resources) are normally scarce relative to the ends that people want to use them for, the use of particular means towards one end often creates 'opportunity costs' by way of foregoing benefits that otherwise could have been obtained by devoting them to other ends. The concept of 'marginal analysis' complements that of opportunity cost. It proposes that any decision should be made on the basis of the incremental benefits and costs expected to result from that decision. Hence, an option should be pursued only when its marginal benefits are expected to exceed its marginal opportunity costs. Marginal analysis within neoclassical welfare economics has been criticised for presuming that the effects of any marginal change are invariably continuous, when effects of environmental decisions not uncommonly are discontinuous (e.g., where loss of a keystone species in an ecosystem 'flips' the ecosystem into another state) (Gowdy et al. 2005). This presumption is not necessary for marginal analysis, however, which can (albeit with considerable modesty) anticipate and account for effects of decisions that are discontinuous

Hajkowicz et al.(2000 p. 45) emphasised as follows the advantages for environmental decision-making of the economic way of thinking:

> Whether we like it or not, we have to make choices about the resources we allocate to environmental projects or the decisions we make about the control of environmental risks. The benefit of economics is that it makes decisions explicit, even though it may be painful to say we cannot afford to invest a certain amount to achieve an environmental objective. It allows the tradeoffs we will make, or may have to make, to be viewed by the public.

The conventional BCA approach to economic evaluation employs a particular methodology for identifying the optimal solution for any given problem. As discussed above, aspects of this methodology are ill-suited to the task of priority setting under collaborative, community-based environmental governance. Moreover, the theoretical foundations of neoclassical welfare economics upon which this approach relies are now subject to serious criticism from within the mainstream of the economics profession. Hence, it has become increasingly difficult to sustain claims for the superior rigour of BCA, compared with alternative or complementary approaches including multiple criteria analysis, that generally were readily accepted between WWII and the 1970s.

Given the ascendancy of neoclassical economics between WWII and the 1970s, and Friedman's (1953) then-persuasive argument that the assumptions of economic theory did not need to be realistic provided the theory could be used to make accurate predictions, most economists during this time remained satisfied with neoclassical welfare economics. With the energy price shocks of the 1970s, and the episodes of global financial instability from the 1980s onwards, however, this satisfaction began dissipating (Gowdy et al. 2001). Although neoclassical welfare economics continues to dominate economic policy discourse around the world, it finds itself in a state of crisis where 'the most serious challenges to the standard welfare paradigm are coming from within the professional mainstream' (Gowdy et al. 2005 p. 208). These challenges have arisen from the:

> … dismantling of its two fundamental pillars (1) the theory of human behaviour embodied in the axioms of consumer choice, and (2) the theory of production embodied in the notion of perfect competition and the marginal productivity theory of distribution. … [Both these concepts] are necessary foundations for neoclassical welfare economics and Pareto efficiency. Neoclassical theorists have by and large abandoned economic man and perfect competition; however, the policy recommendations of economists are still based on these outdated representations of human behaviour and commodity production (Gowdy et al. 2005 pp. 207-208).

Meanwhile, growing numbers of academic and professional economists are turning to more pluralistic methodologies that seek to deal systematically with at least the most important complex elements of problems. The prominent historian of economic thought David Colander (2000 p. 125) predicted at the turn of the millennium that his colleagues of the future would look back on that year as the 'end of the neoclassical era and the beginning of the New Millennum Era'. He predicted also that this shift would mark a transition from the neoclassical 'right price' view of policy to a 'right institutions' view of policy, with this new view analysing the economy as a complex system rather than as the mechanistic system assumed in neoclassical economics. The appropriate approach of economists to complex policy problems, he observed, would be 'loose-fitting pragmatism' rather than deduction of policy prescriptions from first principles (ibid. p. 128). Besley (2007 p. F540) characterised the 'right institutions' approach of economists to policy issues as the 'new political economy', and reflected that this approach 'focuses less on picking good policies *per se*, and more on picking institutions apt to implement and sustain good policies' (ibid. p. F541).

The different methods available for evaluating environmental policy decisions have come to be classified by various schemes. In their review of decision support methods for natural resource management in Australia, Hajkowicz et al. (2000) distinguished analytical techniques (including benefit-cost analysis and multiple criteria analysis) from policy frameworks (including citizens' juries, social impact assessment, and environmental impact assessment). In their view, '[a]nalytical techniques involve a highly specialised, repeatable and structured process for identifying an optimal or 'best' decision alternative' (ibid. p. 113, while 'policy frameworks are much more loosely structured' (ibid. p. 114).

When considering an appropriate decision support tool for the Western Australian Salinity Investment Framework, Black et al. (2002, 2004) argued that the former authors had incorrectly classified the citizens' jury method as a policy framework and had also overlooked related deliberative methods like consensus conferences and deliberative polls. They proposed that it is 'much more logical and helpful' to distinguish methods according to whether they are (a) economic, (b) multiple criteria, or (c) deliberative methods (ibid. p. 162). By 'economic' methods they meant methods conventionally associated with benefit-cost analysis.

Vatn (2005) adopted a similar classification. Recognising that economists are increasingly employing multiple criteria methods, however, he referred to 'benefit-cost analysis' instead of 'economic' methods. His classification of evaluation approaches – into methods of benefit-cost analysis, methods of multi-criteria analysis, and deliberative methods – is followed in the remainder of this document.

1.7 Focus and outline of this document

The purpose of the literature review presented in the remainder of this document is to lay the foundations for choosing in the present project an economic method that is, as discussed above, (i) consistent with government objectives in adopting a community-based approach to regional NRM delivery, (ii) feasible and cost-effective to apply given the capacities of community-based organisations, and (iii) consistent with the 'economic way of thinking'. Accordingly, the document proceeds to review the relevant literature in order, firstly, to identify options for an economic method appropriate to the context in question, and, secondly, to make a preliminary assessment of the relative merits of each option given this context.

The remainder of this document is structured as follows. The approach of benefit-cost analysis to economic evaluation is described and reviewed in chapter 2. The approach of multi-criteria analysis is

described and reviewed in chapter 3, while chapter 4 presents a description and review of deliberative methods.

The development and details of the Investment Framework for Environmental Resources (INFFER) are discussed in chapter 5. This particular framework was singled out for review because it was designed specifically to be applied by community-based environmental organisations, and because it has found significant levels of endorsement and adoption by governments and community-based bodies in Australia. The process of developing this framework also provides useful insights into the political economy of designing a method for economic accountability with realistic prospects of accommodating the diverse needs of stakeholders in community-based environmental governance. Chapter 6 completes the document by summarising its implications for choosing procedures for economic accountability under collaborative, community-based environmental governance.

2. BENEFIT-COST ANALYSIS

Benefit-cost analysis (BCA) is the method of comparing decision alternatives, or setting priorities, conventionally preferred by economists. Many readers of this document will be familiar with this method, which has been explained clearly many times before (e.g., Sinden et al. 1995; Department of Finance 1991; Hajkowicz et al. 2000; Mishan et al. 2007). A brief overview of this method is provided in section 2.1. A range of issues that have arisen in respect of application of this method to environmental decisions are reviewed in section 2.2. Closing remarks are presented in section 2.3.

2.1 Rudiments of BCA

BCA aims to determine whether the welfare of a given public (e.g., nation, regional community) will be, or has been, enhanced as a result of a planned change (e.g., investment, policy adjustment, institutional reform). Where multiple planned changes are under consideration in respect of a particular issue, the method seeks to identify which of the possible options (including non-adoption of any of the other options) would maximise the relevant public's welfare.

Proponents of BCA point to its strong foundations in neoclassical welfare economics as giving it a decisive edge over other methods of priority setting. The advantage of these foundations, it is argued, is that all BCA procedures are based on a common model of economic behaviour that is itself derived from an explicit philosophical foundation. For instance, Bennett (2005 p. 258) noted critically that 'MCA [multiple-criteria analysis] and input-output analysis involve departures from the rigour of welfare economics'.

The philosophical foundations of neoclassical welfare economics centre on the assumption of 'methodological individualism', reflected in the principle of individual sovereignty. The assumption is that individuals know their own preferences with regard to goods and services in a market, and that these preferences are fixed, fully informed, and cannot be influenced by others. The satisfaction of individuals' preferences brings utility, measurable in monetary amounts. These monetary amounts, when summed over all individuals concerned, are taken to comprise 'social welfare'.

Based on these starting assumptions, a sequence of stages akin to the following is generally followed in conducting BCA (adapted from Hajkowicz et al. (2000) and Sinden et al. (1995)):

1. *Define the options, including the 'without project' scenario* (see stage 2).

2. *Identify and quantify the effects of each option that will lead to benefits and costs during the planning horizon of the analysis*. The impacts of the change options are measured relative to the without-project scenario. The without-project scenario should account as realistically as possible for the 'spontaneous' changes that would occur without implementation of the options for planned change (projects) that are under consideration. In most BCA studies, the planning horizon typically extends for 20 to 30 years from the date of the analysis.

3. *Estimate the benefits and costs of each option for each year of the planning horizon*. Where the effects of options under consideration are priced in markets, and where the relevant markets are not significantly affected by 'market failures' (e.g., monopolies or externalities), the effects are translated into monetary benefits and costs on the basis of their relevant market prices. When significant market failures are apparent in respect of a particular effect, the preferred solution is

to estimate what the market price for that effect would be in the absence of those failures (i.e., its 'shadow price'). When a significant market failure is evident in respect of a particular effect but it is not feasible to estimate a shadow price (e.g., for reasons of cost or lack of expertise), the recommended procedure is to acknowledge the effect has been left unaccounted for and to advise decision makers that they need to consider this effect in addition to the BCA results before making a final choice.

4. *Convert streams of benefits and costs into present values*. People normally prefer to receive benefits sooner than later, and incur costs later than sooner. Economists refer to this phenomenon as 'time preference', and account for it in BCA by 'discounting' benefits and costs according to how far into the future they are expected to arise. The present value of the benefits (costs) of an option is calculated as the sum of the discounted annual benefits (costs) of the option. Sinden et al. (1995) recommended that future benefits and costs be discounted at the rate of 5 per cent per year. This is a 'real' rate which assumes benefits and costs are estimated using prices unadjusted for the future effects of price inflation.

5. *Calculate the net present value (NPV) for each option* (other than the without-project option). This is calculated as the difference between the present value of the benefits of the option and the present value of its costs. An option is conventionally deemed economic if its NPV exceeds zero. This criterion accords with the potential Pareto improvement criterion for economic efficiency discussed in section 1.4. When more than one option is under consideration, and there is a budget constraint on the choice of options, the options chosen should be those that maximise aggregate NPV subject to that budget constraint. Where the budget constraint applies to all the costs of the various options, the procedure to be followed in maximising aggregate NPV subject to this constraint starts with calculating a benefit-cost ratio (BCR) for each option. This equals the present value of an option's benefits divided by the present value of its costs. The options are then ranked in descending order of their BCRs. The analyst moves down this ranked list, including options until the budget constraint becomes binding. The NPV-maximising combination of discrete options contains all options included before the constraint becomes binding.

6. *Perform sensitivity testing*. The numeric values used in BCA to estimate the effects of options and translate these into benefits and costs are normally 'best-bet' values. However, there is likely to be considerable uncertainty about some of these values. It is important then to test the robustness of the BCA findings when key values subject to uncertainty are varied across a realistic range.

2.2 Issues with BCA

2.2.1 Declining use and influence

Despite BCA remaining the method conventionally advocated by economists for evaluating public decisions, Watson (2007 p. 6) observed that '[t]echniques like benefit-cost analysis … seem to have gone by the board in the Australian public sector', and Hajkowicz et al. (2000 pp. 45-46) argued that 'economists [in Australia] need to understand why economics is often ignored as a vehicle for advising decision making …'.

There is no one reason for the declining use and influence of BCA in Australian public decision-making, including in respect of environmental decisions. A number of issues that have been identified as contributing to this decline in environmental and natural resources arena are discussed below.

2.2.2 Concerns over assigning monetary values for environmental effects

BCA presumes money to be a neutral measure of value, by which all things can be compared and trade-offs calculated. This presumption has been taken as value free on the basis that money has acquired a dominant role in coordinating economic activity and has thus come to be seen as a fundamental reality (Spash 2008). However, not everyone is comfortable with using money as a single metric for valuing all effects of decisions. Some find it unreasonable to attribute monetary values to effects they regard as intangible, such as benefits from preserving threatened species or saving human lives. Part of this disquiet derives from fears that monetisation of intangible values will erode those values. Rewarding friendship with cash, for instance, may demean the relationship. A review of the use of economic methods in Australia for estimating environmental values on a monetary basis found that '[t]here appears to be a considerable level of skepticism among decision-makers and the community at large about the validity of 'putting a price on the environment' and the results of such studies are treated accordingly' (Government of South Australia 1999 p. 6).

Spash (ibid. p. 274-275) found that such concerns point 'towards the need for research into the appropriate approaches by which plural values held within a given society may be recognised and protected as judged necessary'. In contrast, conventional economists tend to dismiss concerns of this kind as irrational and persist with monetary valuation by way of arguments satisfying their own standards of rationality. For instance, Farquharson et al. (2007) stated:

> The use of money [in economic valuation of environmental changes] is sometimes a barrier to wide acceptance. Many people believe that some environmental assets are 'priceless' in the sense that they cannot accept trade-offs involving these assets, or they consider it immoral to place a value on goods such as clean air or water, which are generally seen as a right for all (Ackerman et al. 2004). However, trade-offs are made all the time with regard to environmental resources, all valuation does is make the extent of the trade-offs explicit.

The risks of such persistence for the public acceptability, use and influence of BCA are highlighted by Nelson's (1987) observation that many participants in policy debates view economists as advocates for a set of values that conflict with their own values, and the warning of Hajkowicz et al. (2000 p. 46) that 'economists must not seek to easily dismiss concerns that are considered to be irrational by economic standards, when for most people they may be a reality'.

2.2.3 Equity concerns

We saw in section 1.4 that (i) social welfare is measured conventionally in BCA according to the potential Pareto improvement criterion, (ii) this criterion is based on the value judgement that the marginal utility of any given amount of money is the same across all individuals irrespective of how rich or poor they might be, and consequently (iii) the incidence of benefits and costs of a change has no effect on BCA's conventional measure of social welfare.

This measure is sometimes defended as democratic on the basis that individuals express the strength of their commitment for any good by the 'dollar votes' they are willing to allocate in acquiring that good. However, ability to express commitment in this way is limited by each individual's income and/or

wealth. The potential Pareto improvement criterion can therefore be regarded as inequitable to the extent that economic inequality exists between individuals in the relevant society. Spash (2008 p. 266) highlighted as follows the inequity of applying this criterion to environmental decisions at the level of global society:

> Several billion people live on less than a few dollars a day, many with a tight subsistence relationship to the ecosystems around them. The environment is an immediate means of survival, and environmental damage has immediate consequences for their and their children's health. Yet these people have little disposable income to express the importance of these systems or their functions. One Bill Gates has far more power in the market place than a few billion people with no income to spare.

Joubert et al. (1997 p. 126) observed similarly that the potential Pareto improvement criterion is 'biased in favour of projects that benefit the already affluent, who are both more willing and more able to 'compensate' those who have lost utility'. Hajkowicz et al. (2000 p. 41) noted in the Australian context of natural resource management that the value judgement underlying this criterion is not universally accepted, and that 'the equity and applicability of BCA is questioned strongly' when this value judgement is inconsistent with community ideals. In considering the choice of a decision support tool for the Western Australian Salinity Investment Framework, Black et al. (2004 p. 159) observed that '[i]n practice, the losses of a lower income group may be weighted more heavily in a policy decision'.

A response by some economists to such concerns is that decisions about what programs or projects to invest in are typically blunt instruments for achieving the equity objectives of a society, and that there are likely to be less costly ways of achieving these objectives than by moving decisions away from those that maximise the society's total net income. Alston et al. (1995 p. 81) found this reasoning 'compelling' in respect of agricultural research decisions, and Dumsday (2001) argued similarly that it is normally more efficient to rely on the social welfare and taxation systems to address equity concerns. Nevertheless, Alston et al. (1995 p. 82) acknowledged that program-level investment decisions sometimes have profound equity implications, that policy instruments to correct for negative equity implications of investment decisions are not always available or applied appropriately, and that in such cases 'a blunt instrument may be better than none'.

2.2.4 Concerns over the validity and affordability of using non-market valuation methods to value unpriced environmental effects

A large part of the reason for environmental problems is that, in many cases, much of the effect of depleting or replenishing environmental assets or natural resources (either as 'sources' of inputs or 'sinks' for wastes) remains external to market transactions and therefore unpriced. In turn, the use and influence of BCA in addressing environmental problems is limited considerably by controversy over the 'non-market valuation' (NMV) methods used conventionally to estimate shadow prices for such environmental externalities[2]. Before proceeding to consider these controversies, it may be useful to describe briefly some of the main NMV methods.

[2] Conventional economists have drawn considerable support in this controversy from environmental scientists who have embraced 'a broadly defined economic approach in the apparent belief that this is a pragmatic solution to the neglect of their principled concerns over the loss of wild Nature and biodiversity' (Spash 2008 p. 261).

2.2.4.1 Methods for non-market valuation

NMV methods can be broadly categorised into three types: (i) market-value methods, which use observable market data for prices to estimate environmental effects of decisions; (ii) revealed-preference methods, which rely on people's preferences for environmental goods being revealed in markets for surrogated good; and (iii) stated-preference methods, which depend on people stating their preferences in response to structured questionnaires (Hajkowicz et al. 2000).

Various market-value methods have been used in valuing environmental externalities (Sinden et al. 1995).

- Defensive-expenditure method. This method recognises that parties often undertake expenditures to defend their existing situation against a negative impact. Provided the defensive action maintains the utility obtained under the prior situation, the cost of the defensive action is taken as a measure of its net social benefit.

- Change-in-cost method. Implementation of a decision may increase or reduce existing costs that can be measured using market prices. Where an action (e.g., adoption of conservation practice) reduces costs, the benefit of the action can be estimated as the costs saved.

- Change-in-output technique. This method is useful when market prices do not exist for the environmental effect to be valued, but they do exist for an output associated with that effect. In such cases, the value of the environmental effect can be deduced from the change in the revenue from the associated output. For instance, the benefit from soil conservation can be estimated using this method as the increased revenue from the additional agricultural output that results.

- Replacement-cost method. The amount that a community actually pays (or is willing to pay) to replace an existing benefit that has been (or is expected to be) lost is a measure of the minimum level of that benefit. The amount is a minimum estimate because a rational community would not incur the replacement cost unless that cost was less than the lost benefits to be replaced.

The category of revealed-preference methods also includes a number of methods, including the two following (ibid.).

- Travel cost method. On the basis of data on (a) visitation rates to an amenity (e.g., a national park) from zones differing in their distance from the amenity, (b) the costs of traveling to the amenity from each of the zones, and (c) other variables expected to affect visitation rates, a demand schedule for amenity visits is calculated. From this schedule, society's benefits from the amenity (measured by aggregate willingness to pay for visits to the amenity) is estimated.

- Hedonic pricing. This method of estimating the social benefit obtained from unpriced goods follows from the observation that a good is actually a bundle of characteristics, and it is sought for the utility that each of the characteristics provides. Sinden et al. (ibid.) illustrated this method using the much-simplified example of two houses, A and B. The price of A is $50,000 higher than the price for B. All the characteristics of the houses are identical, except A is located further from a source of heavy air pollution. The cost of the air pollution for the owner of B is thus calculated as the difference in house prices, which is $50,000.

A number of stated-preference methods have also been used (ibid.).

- Contingent valuation. In this method, the value of an unpriced effect of a decision is estimated by asking individuals to nominate the maximum price they would be willing to pay to obtain it (if it's a positive effect) or avoid it (if it's a negative effect). The response from each individual provides an estimate of the total benefit that they expect from the effect. The individual's net benefit (consumer's surplus) can then be calculated by deducting from their total benefit the costs that they would incur in obtaining that benefit.

- Choice modelling. While contingent valuation usually requires respondents to choose between a base option and a single alternative, choice modelling requires respondents to choose between different goods made up of unique bundles of attributes (including environmental attributes and a price for using or accessing the good). By choosing between different goods partly on the basis of the price attribute, respondents reveal the monetary value of the other attributes indirectly.

The controversy about NMV methods has been partly due to the concerns over monetising environmental values and over the equity implications of conventional BCA that were discussed in sections 2.2.2 and 2.2.3, respectively. It is also due to other concerns, however, and these additional concerns are addressed in the remainder of section 2.2.4. The controversy is such that Adamowicz (2004 p. 438) found that '[w]hile there continues to be significant interest in academic research on [non-market environmental valuation], the policy implementation of this work is not as active as one would hope'.

Conventional economists focused on environmental issues (here called 'environmental economists') acknowledge a number of these additional concerns and the need to address them. For instance, Bennett (2005) noted the ongoing debate within the Australian environmental economics profession regarding the validity of shadow prices estimated by NMV studies. Prominent in this debate were concerns over the potential for strategic bias arising from the hypothetical nature of questions used in contingent valuation (CV) studies. He remarked that the resulting controversy had made environmental policy advisers cautious in commissioning such studies and strengthened the hand of decision makers and lobbyists in questioning the usefulness and validity of including NMV estimates within BCA. Bennett observed how the controversy spurred further development of the CV method and provided part of the impetus for developing choice modelling (CM) as a stated-preference method less prone to strategic bias. Nevertheless, he observed that 'the CV validity debate has enveloped CM …' (ibid. p. 251).

2.2.4.2 Disagreement among environmental economists on the requirements of neoclassical welfare economics

Meanwhile, disagreements between leading Australian environmental economists regarding the relative merits of particular NMV methods have added to the reasons for environmental policy makers being cautious in commissioning application of NMV methods generally and basing decisions on their findings. For instance, Bennett (2005 p. 254) commented in a section headed 'cop-outs' on a tendency among decision makers, or at least those pressed into obtaining estimates of non-market values, to commission market-value methods like the defensive-expenditure method or the replacement-cost method that 'are inconsistent with principles of welfare economics'. He illustrated the replacement-cost method with an example where 'the costs associated with draining and developing a wetland may be estimated using the replacement costs method that involves the calculation of costs associated with the

construction of an artificial wetland'. He considered this approach 'problematic because the artificial wetland may not be a complete substitute for the original' and 'the cost of the replacement has no conceptual link to the net benefit enjoyed from the original wetland' (ibid.). He criticised the defensive-expenditure method for similar reasons, and concluded that '[a]pplying either of these techniques has the potential to deliver erroneous policy advice, an ironic consequence of attempts to avoid errors arising from the use of stated preference techniques' (ibid.).

In turn, Sinden et al. (2008) argued that NMV methods based on actual cost data, such as the defensive-expenditure method, better satisfy the principles of neoclassical welfare economics than do stated-preference methods. In their view, these principles require non-market values estimated for environmental services to be the values that would occur in a competitive market for those services. They argued accordingly that any NMV method is appropriate only to the extent that it models the structural characteristics of such a market, including (i) many buyers, all facing budget constraints; (ii) many producers, all facing input constraints; and (iii) exchanges between buyers and sellers. They observed that stated-preference methods are deficient by this standard, and actual-cost-based methods superior, since the former 'have no effective budget constraints, whereas the methods based on actual costs do' (ibid. 6)[3]. Moreover, they concluded from a review of Australian stated-preference valuations of biodiversity benefits that the values generated from such studies 'sometimes lack consistency' (ibid. p. 2).

2.2.4.3 Appropriateness of the principle of individual (consumer) sovereignty

We saw in section 1.4 that conventional methods of economic evaluation, including BCA and associated NMV methods, adhere to the principle of individual (or consumer) sovereignty. Accordingly, the independently-formed preferences of individuals are preferred as the basis for estimating the aggregate (social) welfare effects of a given decision. The aggregate effect on social welfare is thus calculated as the sum of the welfare effects on individuals as interpreted through the lens of their own preferences. This approach is known as methodological individualism.

The appropriateness of this principle as a basis for estimating non-market environmental values has come under sustained questioning. Part of this questioning relates to environmental economists committing to this principle on the basis of an assumption that individuals' preferences exist prior to a choice so that they know what they want. This assumption has been criticised as a basis for valuing environmental effects of decisions that few individuals other than experts would actually understand. Hajkowicz et al. (2000 p. 21) observed:

> Many ecologists are puzzled that whilst it is unknown which species play a redundant role and which play a keystone role in most ecosystems, many economic studies appear to assume that consumers, aided by the market place, can correctly place a value on various species, etc., even though many respondents could not even identify them.

Spash (2008 p. 270) remarked similarly:

> Describing and understanding ecosystems functions requires alien concepts divorced from daily life. There is then a disconnect between the 'goods' demanded by the public and ecosystems services derived as outputs from functions conceptualised by ecological science.

[3] Sinden et al. (2007) argued similarly.

A particular consequence of respondents in stated-preference NMV studies being ill-informed about the ecosystem effects of decisions is that respondents tend to be more interested in species that give them 'warm and fuzzy' feelings. Hence, '[t]he focus of economic valuation studies has … been upon key iconic species, and so far has rarely addressed species diversity, and hardly ever ecosystems and never genetic diversity' (ibid. p. 270).

Often faced with ill-informed respondents in valuing the ecosystem effects of decisions, environmental economists have tended to assume that stated-preference NMV methods can be applied such that respondents' ignorance can be corrected by providing them with information without affecting their preferences. However, available evidence demonstrates that the same set of information on concepts like biodiversity will inform some individuals but influence the preferences of others (Spash 2002).

Given these challenges of applying methodological individualism to environmental valuation, Spash (2008 pp. 276-277) remarked that:

> The problem facing those ecologists promoting ecosystems services valuation is that most of what they deem valuable is unlikely to produce meaningful willingness to pay amongst the general population, e.g. using stated preference methods. Undaunted, studies place monistic monetary values on changes in a large number of ecosystem functions involving everything from nutrient recycling to cultural heritage. In order to achieve these numbers, the evaluator freely borrows and transfers values from a variety of economic studies with little apparent consideration of the original context or theoretical basis of those values. … Meanwhile, the overall approach is justified as pragmatic, ideologically sound, empirically based and even democratic.

2.2.4.4 Lack of opportunities for deliberation over non-market values

Alongside concerns about perfectly-formed preferences and the principle of individual sovereignty as a basis for valuing complex environmental effects have been another criticism of conventional NMV methods. This criticisms is that such methods leave no room for individuals to engage in deliberative discourse with one another to share information and move towards agreement on a collective value system against which values of particular environmental effects can be directly measured (instead of having economists derive their collective value system for them indirectly, by summing their prior preferences). Deliberative discourse relies on 'establishing conditions of free public reasoning among equals who are governed by the decisions' (Cohen 1998 p. 186).

The noted environmental economist Alan Randall (1999 p. 32) recognised in environmental policy problems that '[s]tructured discourse and deliberation can often undermine conflict, and careful consideration of information can erode firmly held priors and open up new possibilities'. While the conventional approach to economic evaluation would regard such deliberation as breaching individuals' privacy by opening the door for paternalism, Norton et al. (1998) found it mistaken to view efforts to influence preferences as paternalistic when such efforts follow rules decided democratically by the participants. They argued further that deliberative processes of 'democratic preference change' offer individuals opportunities to counterbalance the preference-changing agendas of vested interests (e.g., as pursued through public relations campaigns) and thereby gain greater sovereignty over their preferences than they could achieve independently.

In response to criticisms of conventional environmental NMV methods for assuming respondents have well formed and informed preferences, and also for excluding various sustainability issues like rights and fairness, some economists (mainly ecological economists) have proposed and explored alternative

approaches. Much of the emphasis of ecological economists (e.g., Funtowicz et al. 1990; Meppem et al. 1999) has been on enhancing quality in the social process of valuation, with deliberative and participatory fora seen as a way of continuing to help decision makers while avoiding the need for NMV studies and BCA more generally.

Following Sagoff (1998 p. 213) who argued for the NMV method of contingent valuation becoming 'a deliberative, discursive, jury-like research method emphasizing informed discussion leading toward a consensus based on argument about the public interest', some economists (e.g., Niemeyer et al. 2001; Macmillan et al. 2006; Howarth et al. 2006; Lienhoop et al. 2007) have begun exploring the scope for 'deliberative monetary valuation' (DMV) as a way of combining BCA with deliberative and participatory methods. The aim of DMV is to estimate monetised social values directly rather than by estimating individuals' values and then aggregating these. After reviewing the work in this area, Spash (2007 pp. 697, 698) found this approach 'to raise more questions than it answers. … In using DMV economists have landed themselves squarely in the middle of the philosophical debate over realms of value and the political debate over representation of different values in society'. Moreover, he found that social values estimated in DMV studies varied according to the institutional setting and the particular process of valuation.

2.2.4.5 Affordability of NMV methods

BCA analysts concerned with environmental problems need to 'practice what they preach' and ensure that the costs of their analyses do not exceed the likely benefits from enhancing the quality of decision making. Schmid (1989 p. 287) remarked accordingly that 'BCA's benefits can outweigh its costs, but for that to be true, the analyst must be modest and cognizant of data and decision costs'.

This constraint on BCA analysts is real enough when most effects of the decisions at issue can be valued using market prices, but it becomes appreciably more pressing when evaluating decisions for which some of the most important effects are non-market environmental ones for which conventional BCA requires valuation using NMV methods. Application of most such methods requires sophisticated skills (usually in short supply) and often also involves costly processes of data collection. Hajkowicz et al. (2000 p. 24) observed in the Australian context that '[t]he application of environmental valuation techniques is expensive; it can consume large amounts of time and resources', and Black et al. (2004 p. 163) noted that 'it is often the case that the resources or time are not available to apply [non-market valuation techniques] to all possible impacts under consideration'.

For instance, Farquharson et al. (2007) reported that the cost of undertaking new NMV studies using choice modeling is in the order of $A100,000 to $A140,000, depending on the kind of survey method used. They found that '[o]nly highly contentious cases where large values are involved will warrant direct data collection exercises [in applying NMV methods]' (ibid. p. 9). Similarly, Hajkowicz et al. (2000) argued that BCA analysts should apply NMV methods only to those environmental effects whose monetary valuation would cost less than the expected decision-making benefits, and account for other environmental effects by applying benefit-transfer techniques or the threshold-comparison approach.

Bennett (2005 p. 253) attributed the lack of 'integration of NMV into policy-making in both Australia and New Zealand' not only to controversy over the validity of results from NMV methods (see section 2.2.4) but also to 'the relatively high costs of undertaking stated preference techniques'. He was concerned that the cost issue had led decision makers to rely on NMV methods (such as replacement-cost and defensive-expenditure methods) which he regarded as less consistent with the principles of

neoclassical welfare economics (see section 2.2.4.2), and to increased reliance on the benefit-transfer approach which he viewed as having limited present scope for valid application (see below). He was concerned too that the relatively high costs of stated-preference methods, combined with questions over their validity, had led some economists into using 'avoidance strategies' such as multiple criteria analysis which 'involve departures from the rigour of welfare economics. The consequence is that the problems avoided by short-cut analysis result in different and often more intractable problems' (ibid. p. 258). As noted in section 1.6, however, claims for theoretical rigour in the neoclassical welfare economics underpinning BCA and NMV methods nowadays find limited support from mainstream economic theorists.

A number of the techniques mentioned in this section – namely the benefit-transfer method, the threshold approach, and multiple criteria analysis (MCA) – have not been discussed previously and are considered below. MCA is discussed in chapter 3.

Benefit-transfer approach

Wilson et al. (2006 pp. 335-336) noted the increasing demands on agencies to assess the benefits and costs of projects with environmental consequences, and the time and resource constraints faced by such agencies, and remarked accordingly that:

> … it is perhaps not too surprising that the benefit transfer approach has spread steadily in the last few decades as decision makers have sought timely and low cost ways to assign monetary values to goods and services that are not commonly traded in the market place. Conducting original valuation research is time consuming and expensive. Policy analysts are often constrained in their ability to support new research studies within mandated deadlines.

Bennett (ibid. p. 254) defined this approach as involving 'values previously estimated in some case study (the source) being used (transferred) to another setting (target)'. He argued that its scope was limited because the circumstances present in the source study need to resemble those present in the target setting, the source study needs to have involved sound application of a valid technique, and there was a 'paucity of source studies' from which transfers satisfying the prior two conditions could be undertaken.

Nevertheless, Farquharson et al. (2007) identified a number of ways that source-study valuations might be adjusted in an attempt to address these concerns: (i) expert judgement of how a value might vary between the source and target sites; (ii) re-analysis of source-study samples to identify sub-samples of data suitable for transfer; and (iii) meta-analysis of values estimated across various source studies to estimate benefit-transfer functions applicable to particular target sites.

Moreover, van Bueren et al. (2004) and Windle et al. (2007) have undertaken choice-modelling studies to examine the validity of transferring environmental values estimated in one context to another. The authors of the former study concluded that 'it is clear that the challenge of benefit transfer is greatest when source values are required for evaluating welfare impacts at a localised level where values are highly dependent on the context in which the environmental outcomes are embedded' (van Bueren et al. 2004 p. 27). They found that it is problematical to account for all contextual differences between source and target settings in a benefit-transfer function. For instance, it can be difficult to account for differences in the availability of substitutes for the environmental attribute at issue. It can be challenging also to account for differences in cultural characteristics 'such as attitudes and social norms

… [that] have the capacity to influence willingness to pay' (ibid. 4). Adamowicz (2004 p. 440) concluded along these lines that:

> … further development of benefit transfer techniques, especially preference calibration, is very important [for increasing the use on environmental valuation in policy analysis]. Such research will have to include assessments of the degree to which benefit estimates vary by demographic, cultural and other factors.

The latter study involved a series of choice-modelling studies undertaken in Queensland to build a reference data base of values to be drawn upon in benefit-transfer exercises. Its authors found 'it is possible to design an evaluation exercise that will provide a database of values that are broadly relevant and suitable for benefit transfer at the state level' and that 'the marginal values for improvements in resource stocks in Queensland determined in the valuation exercise can be reasonably applied in a range of specific or more general contexts within the State' (Windle et al. 2007 p. 14).

On the basis of these and other studies, Farquharson et al. (2007 p. 16) concluded that a 'judicious approach to [choice-modelling] valuation and the appropriate use of benefit transfer will allow a practical approach to NRM decision making [by Catchment Management Authorities in New South Wales]'. Nevertheless, considerable scepticism persists among economists, and especially among ecological economists, regarding the merits of the benefit-transfer approach. For instance, Spash et al. (2006 p. 387) observed that benefit (or value) transfer has a serious range of caveats typically paid insufficient attention. They reported that '[o]verall results from convergent validity tests show that the uncertainty in value transfers, both spatially and temporally can be considerable. Yet, this is but one test of validity and failing to pass others, such as face and construct validity, is too readily ignored'. They highlighted too 'that including socio-economic, demographic and attitudinal variables in transfer functions may be at odds with the very theoretical foundations of the valuation studies involved, and that this problem seems largely unobserved'(ibid. p. 387). Overall, they judged that 'monetary value transfers are but one, often very imperfect, approach which closes down problems when they may require opening up' (ibid. p. 386).

Threshold-comparison approach

The threshold-comparison approach tends to be the method of last resort for economists committed to applying BCA in the conventional manner. This approach comes into its own when a important non-market effect of a decision alternative cannot be valued because of the complexity involved, the cost of doing so, or practical difficulties including time constraints (Sinden et al. 1995).

When the effect is beneficial and the BCA not accounting for its value finds the NPV of the decision alternative to be less than zero, the threshold-comparison method involves calculating what the value of the unaccounted-for benefit needs to be to increase NPV to the threshold value of zero. This 'break-even' value is then presented to the decision maker/s who is asked to judge whether the actual social value of the benefit is likely to exceed the break-even amount. An affirmative answer is taken as indicating that the alternative would increase economic efficiency.

When the effect is detrimental and the BCA not accounting for its value finds the NPV of the decision alternative to exceed zero, in contrast, the threshold-comparison method involves calculating what the value of the unaccounted-for cost needs to be to reduce NPV to zero. This 'break-even' value is then presented to the decision maker/s who is asked to judge whether the actual social value of the cost is

likely to exceed the break-even amount. An affirmative answer is taken as indicating that the alternative would reduce economic efficiency.

Bennett (2005 p. 251) argued that relying on the threshold-comparison approach to account for non-market environmental effects in BCA studies, instead of relying on NMV studies, 'provides decision-makers and lobbyists with more degrees of freedom to pursue rents in the decision-making process'. This is because decision makers' judgements of whether actual benefits (costs) exceed (fall below) breakeven values are subjective or 'political' and thus open to opportunistic manipulation, whereas NMV studies are viewed by conventional economists as purely technical and therefore free from the risk of such manipulation. NMV studies, like BCA studies more generally, do involve subjective value judgements (see section 1.5), however, and stated-preference NMV studies can also be open to strategic manipulation by respondents (see section 2.2.4.1).

2.2.5 Inability to foster and account for deliberatively-determined value systems

Having discussed a range of reasons for controversy over the use of NMV methods in environmentally-focused BCA studies, we return to considering issues applying to such studies generally. One such issue relates to the discussion in section 2.2.4.4 concerning the lack of opportunity in NMV studies for respondents to deliberate with one another to share information and move towards agreement on a collective value system against which non-market environmental values can be measured. The informational and cognitive constraints faced by such respondents are not dissimilar to those often faced by decision makers seeking advice from economists or other decision analysts. The reasons for such decision makers seeking to alleviate these constraints by engaging in deliberative processes are also not dissimilar to the need for NMV-study respondents to deliberate with each another.

Norton (2005) explained that the value of deliberative discourse in environmental decision making derives from most such decisions having the classic characteristics of problems that Rittel et al. (1973) described as 'wicked'. Each such problem emerges as a 'mess' because different parties cannot agree on what the problem is because their divergent interests lead them to frame it differently. Norton et al. (2007) reasoned that the positivist approach of BCA to economic evaluation, which regards problems and social value systems as 'givens' and therefore objective, is inappropriate when parties to a problem cannot agree on what the problem actually is. Issues that need to be debated openly in a pluralistic manner get corralled into a monistic framework that sets the parameters by which these issues become understood. Consistent with the comment of Spash et al. (2006) in 2.2.4.5 in respect of the benefit-transfer approach, the conventional approach to economic evaluation promptly closes down 'messes' at the cost of foregoing vital opportunities to open them up in order to 'surface' and deliberate upon values than can be agreed upon, and thereby innovate solutions consistent with agreed values and thus more likely to be implemented at an affordable political cost.

We saw in section 1.4 that a key reason for governments sponsoring collaborative community-based processes of environmental management derives from the advantages of such processes for reducing conflicts between parties to a decision that arise from differences in the knowledge and values they apply in defining the problem and evaluating alternative solutions. By moving parties towards agreement on a knowledge system they can apply in defining their problem, and on a value system they can use in ranking alternative solutions, conflicts can be reduced and legitimacy and community ownership of decisions thereby increased. As noted above, however, conventional BCA precludes such

advantages by imposing a collective value system (the potential Pareto improvement criterion) claimed[4] to be value free.

Schmid (1989) understood implicitly that BCA normally encounters wicked problems for which the conventional approach to BCA is counter-productive. He recognised that '[d]ecision makers are not always ready to answer the questions put to them by analysts. In many cases their objectives have not been thought out' (ibid. p. 286). It follows there are advantages not only in different decision makers sharing the same problem deliberating with one another but also in engaging deliberatively with the BCA analyst. Schmid proposed accordingly that BCA proceed as 'a dialogue between analyst and public decision makers. It is not something that the analyst does alone and presents finished to the world … This interactive dialogue is part of the creation of objectives' (ibid. p. 286). The aspiration in conventional BCA is to reduce as far as possible the scope for ad hoc decision making by providing technical solutions to value conflicts that would otherwise invite political judgement. The counter-productivity of this aspiration in the case of wicked problems comes from 'isolated, sweeping, and noninteractive analysis invite[ing] piecemeal, ad hoc decisions as politicians change the analyst's often unintended presumptuous resolution of value conflicts' (ibid. p. 286).

The appropriate role of the BCA analyst faced with wicked problems, Schmid (ibid. p. 286) argued, is to 'raise the necessary questions requiring political resolution of conflicting interests and simulate a range of likely answers'. In doing so, the analyst needs to recognise that '[e]veryone in government is part of the political process, the hired analyst as well as the elected politician. The analyst as a practical matter cannot have every detail … ratified by the legislative body' (ibid. p 289). Given this role, BCA becomes 'a framework for systematically displaying the consequences of alternative spending and regulations in such a manner that the ranking of these alternatives is the result of applying politically chosen rules reflecting explicit performance objectives' (ibid. p. 285). Schmid (ibid. p. 288) called this approach to BCA the 'political economy approach', noting its similarity to the decision-making approaches proposed previously by Peacock (1973) and Sugden et al. (1978). Other economists advocating the approach include Bromley (1989) and Marshall (2005).

Schmid (ibid. pp. 287, 304) recognised as follows that decision makers, and indeed citizens, do not always prefer to be explicit and consistent in respect of the objectives against which they would like choices evaluated: '[P]oliticians confronted with a divided public may prefer obfuscation, and the public faced with difficult choices may prefer escapism … [P]olitical choice of any kind has as one of its ingredients symbolic manipulation including hiding value conflicts behind various smokescreens … that happen to serve particular interests'. Nevertheless, he described his approach as 'an argument for asking more questions of politicians' (ibid. p. 303) and urged BCA analysts not to 'be apologetic for asking questions rather than supplying independently determinative project values and rankings' (ibid. p. 305). As such, he judged political-economy BCA to be worthwhile as part of the process of strengthening accountability and participation in democratic government.

2.2.6 Misidentification of environmental problems as mechanistic

Adamowicz (2004) noted that decisions about problems for which passive (non-use) values of the natural environment (e.g., option, existence and bequest values in respect of preserving endangered species) are important are rarely made in an explicit benefit cost framework. He remarked on the potential for irreversible changes in such values. Given this potential 'and our limited understanding of

[4] Incorrectly, for reasons explained in section 1.4.

values and preferences in the context of irreversibility and dynamics', he concluded that the lack of application of BCA to such problems 'may be quite appropriate' (ibid. p. 427).

Implicitly at least, Adamowicz recognised that ecological systems tend to be complex adaptive systems characterised by dynamics quite different from those assumed to be at work in the neoclassical welfare economics that BCA is based on. This economics assumes that the dynamics of all systems under analysis are mechanistic. A mechanistic system is characterised by constant relationships between constant parts, so that a single constant relationship exists between cause and effect. Any intervention in (or random shock to) a mechanistic system therefore moves it from one state of equilibrium to a unique and predictable new one. Reversing the intervention reinstates the original equilibrium. The transition from an intervention to a new unique equilibrium occurs by virtue of negative-feedback dynamics (e.g., friction in a real machine) which eventually dampen to zero the ongoing effects of the intervention. The negative-feedback dynamics ensuring this outcome in neoclassical welfare economics are called 'diminishing returns' (Marshall 2005).

Many natural and social systems are characterised by positive-feedback dynamics as well as negative feedbacks. Positive-feedback dynamics amplify any intervention in (or random shock to) a system and can thereby 'flip' it randomly towards any one of multiple possible equilibria, or even maintain it in ongoing disequilibrium. Systems with behaviour influenced significantly by such dynamics are called complex adaptive systems. The elements of such systems 'adapt to the world – the aggregate pattern – they co-create. … As the elements react, the aggregate changes; as the aggregate changes, elements react anew' (Arthur 1999 p. 107). Such a system is complex not because it consists of many elements but because its patterns of behaviour are 'emergent' in the sense that they cannot be adequately understood by focusing only on the elements. Unlike a mechanistic system, a complex adaptive system features changing relationships between changing parts, and thus between cause and effect. Hence, complex adaptive systems are subject to irreversibility in the sense that reversing an intervention is unlikely to fully reverse its effect and reinstate the prior state.

Adamowicz (2004) recognised that ecological systems are prone to significant irreversibilities of this kind, and consequently that the mechanistic method of BCA may be inappropriate for problems involving such systems. However, he assumed implicitly that significant irreversibilities of this kind generally occur only in the kinds of ecological systems which economists regard as generating passive values, and that BCA's mechanistic method therefore remains appropriate for estimating all other values. However, a consensus is emerging among economists and other social scientists that social systems often behave more like complex adaptive systems than mechanistic systems, and need to be analysed and managed accordingly (e.g., Arthur 1999; Gallagher et al. 1999; Goldenfield et al. 1999; Colander 2000; Brock et al. 2007; Ostrom 2005; Nelson et al. 2008; North 1990). This trend has been accompanied in the economics discipline by growing awareness of the importance of increasing-return (equivalent to positive-feedback) dynamics for understanding contemporary problems (Arthur 1988; 1989, 1994). The mechanistic approach of neoclassical economics has depended on maintaining the argument that these dynamics are normally unimportant enough for most economic problems that they can be safely assumed away. The growing awareness among economists of the real-world significance of increasing returns has led to increasing attention within their discipline to irreversibilities (or 'path dependencies') arising from policy interventions and the need to account for them in evaluating such interventions. Such economists working on Australian environmental issues have included Challen (2000), Marshall (2003, 2005) and Pagan (2003). Marshall (2005) was concerned particularly with the need to account for the transaction-cost consequences of irreversibilities in the process of developing the social capital needed for collaborative community-based environmental management.

Indeed, the contention that social systems are less complex than ecological systems, and that mechanistic methods therefore remain appropriate when analysing the former, is becoming superseded by arguments that these systems are so closely intertwined that integrated approaches capable of understanding them as a joint complex adaptive system are required. For instance, Holling et al. (1998 p. 352) characterised environmental problems as follows:

> Characteristically, these problems tend to be systems problems, where aspects of behaviour are complex and unpredictable and where causes, while at times simple (when finally understood), are always multiple. … This is true for both natural and social systems. In fact, they are one system, with critical feedbacks across temporal and spatial scales. Therefore interdisciplinary and integrated modes of inquiry are needed for understanding. Furthermore, understanding (but not necessarily complete explanation) of the combined system of humans and nature is needed to formulate policies.

This kind of reasoning has led environmental problems to be defined increasingly in relation to 'social-ecological systems' (SESs). Anderies et al. (2004 p. 3) defined a system of this kind as:

> … an ecological system intricately linked with and affected by one or more social systems. …We use the term 'SES' to refer to the subset of social systems in which some of the interdependent relationships among humans are mediated through interactions with biophysical and non-human biological units. … When social and ecological systems are so linked, the overall SES is a complex, adaptive system involving multiple subsystems, as well as being embedded in multiple larger systems.

This line of thought indicates Adamowicz's (2004) doubts about the appropriateness of BCA may be relevant to many more environmental problems than he intended.

Meanwhile, the evolving public discourse on environmental policy in Australia reflects an increasing awareness that many of the social-ecological systems of concern in this domain behave as complex adaptive systems. This trend is perhaps most evident from the increasing frequency with which the language of 'resilience thinking' (Walker et al. 2006) has come to be found in policy documents[5]. This language and mode of analysis emerged originally from ecologists recognising that many ecosystems are usefully analysed as complex adaptive systems, and that such systems are prone to 'flip' surprisingly into undesirable states when certain thresholds are approached. This recognition led first to a focus on the resilience of ecosystems, and eventually of whole social-ecological systems, with resilience coming to be defined as 'a measure of a system's capacity to cope with shocks and undergo change while retaining essentially the same structure and function' (Walker et al. 2009 p. 1). One example of the influence of resilience thinking on Australian environmental policy lies in the NSW Government's adoption of the following state-wide aspirational goal: 'Resilient, ecologically sustainable landscapes functioning effectively at all scales and supporting the environmental, economic, social and cultural values of communities' (quoted in Natural Resources Commission 2008b p. 9). Another comes from the 2009-2010 Business Plan for the Australian Government's Caring for our Country program which 'seeks to achieve an environment that is healthy, better protected, well-managed and resilient, and provides essential ecosystem services in a changing climate' (Commonwealth of Australia 2008a p. 3).

[5] Even so, Walker (2009 p. 3) remarked recently that '[t]hough references to 'resilience' now appear in various Departmental vision and mission statements, it is not yet being applied or researched in policy development in Australia'.

2.2.7 Difficulty of accounting for transaction costs

Bennett (2005 pp. 256, 258) commented on the increasing interest in developing market-based instruments for environmental management, and that:

> One element that has been neglected in this field is the empirical assessment of transaction costs[6] under alternative institutional settings. … Inclusion of the analysis of transaction costs will be important to ensure completeness in developing policy advice. Given that transaction costs have been found to account for more than half of all the costs of producing and distributing the national product in modern market economies[7] (North 1990), their omission from the development of extensions of the market to encompass environmental resources could be serious.

Nevertheless, accounting for transaction costs in BCA studies of environmental issues has proven difficult. McCann et al. (2005 p. 538) concluded that '[t]ransaction costs have yet to be fully operationalized in a neoclassical framework, which hinders comparative policy evaluation'. They reported that '[m]ost of the transaction cost measurement studies to date use words like 'crude' or 'approximate' to qualify their results. However, these studies have demonstrated that measuring transaction costs is important because the magnitudes, particularly for nonpoint pollution policies, are significant' (ibid. p. 539. A major constraint on progress in this regard is the frequent lack of collection of transaction cost data by public agencies, and these authors recommended that collecting such data 'should become a routine part of public agency activities to increase efficiency since 'what gets measured gets managed'' (ibid. p. 539). Even so, they noted that the costs of this data collection can be substantial and need to be justified by the likely decision-making benefits.

If accounting for transaction costs under the mechanistic method of neoclassical welfare economics is difficult, it is much more difficult again when the problem at hand involves a complex social-ecological system. When this is the case, such costs will be influenced significantly by increasing-return dynamics which the neoclassical method regards as unimportant. Challen (2000) argued that such dynamics can influence strongly the total transaction costs ultimately arising from environmental policy or institutional decisions[8]. He was concerned especially with how such decisions and dynamics result in new patterns of vested interests which resist reversal of these decisions can thereby create 'institutional path dependence'[9]. Accordingly, he proposed a framework for comparing the cost-effectiveness[10] of institutional alternatives which accounts not only for the kinds of transaction costs recognised by the

---

[6] Furobotn *et al.* (1992 p. 8) regarded transaction costs as '... most easily understood as embracing all those costs that are connected with (i) the creation or change of an institution or organisation, and (ii) the use of the institution or organisation'.

[7] Falconer *et al.* (2001) found that for each £1 paid to farmers from 1995/96 government expenditures to undertake on-ground actions under English agri-environmental schemes, transaction costs of £0.41 were incurred running the schemes and monitoring compliance.

[8] Challen's particular focus was water policy within the Murray-Darling Basin.

[9] Challen took his lead from Dixit (1996 p. 26) who observed how '[p]olicy acts shape the future environment by creating constituencies that gain from the policy, who will then fiercely resist any change that take away those gains'. He was influenced also by North (1990) who argued that such path dependence in policy choices arises not only by creating new vested interests but also by affecting what people do and learn from, and thus by affecting their mental models and what they actually learn.

[10] The basic steps in applying cost-effectiveness analysis (CEA) are the same as for BCA, except CEA does not require monetary measurement of benefits. CEA is typically applied when it is not possible to estimate the benefits of all decision alternatives as required for BCA. Alternatives are ranked according to their monetary cost (in present value terms) per unit of physical effectiveness (e.g., dollar cost per hectare of native vegetation conserved). CEA is well suited to situations where a fixed (or substantially fixed) amount of resources is already committed for expenditure, leaving to be decided only the question of how this amount should be expended (Sinden et al. 1995).

neoclassical method but also for a further category of transaction costs ('intertemporal opportunity costs') that arise from this type of path dependence.

Path dependence follows from the increasing-return (or positive-feedback) dynamics of complex adaptive systems (Arthur 1999). Hence, those transaction costs arising from institutional path dependence are emergent and cannot accurately be deduced using a mechanistic method. Rather, the appropriate method for identifying emergent effects within a complex adaptive system is inductive, which involves:

> … look[ing] for patterns; and we simplify the problem by using these patterns to construct temporary hypotheses to work with. We carry out localized deductions based on our current hypotheses and act on them. As feedback from the environment comes in, we may strengthen or weaken our beliefs in our current hypotheses, discarding some when they cease to perform, and replacing them as needed with new ones (ibid. p. 406).

Challen (2000 p. 207) recognised accordingly that applying the conventional economic approach in a setting of institutional path-dependence is 'impeded by a lack of techniques and methodology for *ex ante* estimation of transaction costs'. Marshall (2005, 2003) adapted Challen's framework for cost-effectiveness analysis (CEA) to account also for the effects of institutional decisions on transformation (production) costs. Institutional choices affect transformation costs as well as transaction costs, and it is quite possible for the transaction-cost advantages of an alternative to be outweighed by its transformation-cost disadvantages (McCann et al. 2005). Marshall (2005) proposed a research strategy whereby economists might develop some capacity for inductive prediction of costs of institutional decisions that arise emergently from path dependence. He cautioned that 'the predictions we can hope to make will at best be rough and highly contingent on how random events unfold' (ibid. p. 146).

2.2.8 Practicality and affordability of BCA for disaggregated decisions

Farquharson (2007 p. 1) made a case for applying BCA in NSW under the regional delivery model 'as an alternative means of assessing *ex ante* questions of priority setting at the catchment [regional] level and for project appraisal'. They argued that using BCA to evaluate which project bids to fund would be superior to the scoring, ranking and indexing (referred to as 'non-economic') approaches typically applied by Catchment Management Authorities in NSW. Among their concerns with these approaches was 'they are not based on any theoretical construct of society's objective function. The choice of criteria to score or rate projects is in the hands of technical experts' (ibid. p. 7).

This call for strengthening the economic accountability in the project-level funding decisions of regional NRM bodies through increased use of BCA brings to mind the Australian experience during the early to mid 1990s when some public funding organisations for agricultural research and development (R&D) (e.g., Grains Research and Development Corporation) required each project funding proposal to be supported by an *ex ante* BCA. While most agricultural economists were initially enthusiastic about this move, many involved in the process became sceptical of its merits. Prominent among these was Pannell (1996, 1997). Some of his arguments against making project-level BCA compulsory in the R&D-funding sphere are relevant for the NRM-funding sphere.

One argument related to the feasibility of performing rigorous BCA studies for numerous project-level R&D proposals given the normally tight timelines and the relative lack of personnel with the requisite skills and experience:

> Even where appropriate, the BCAs are generally of low quality. Given the time frame within which the BCAs are typically done for the Corporations, it is not possible even for experienced analysts to produce high quality results for most proposals. The lack of expertise of many of the people actually conducting the analyses (whether they be biological scientists or junior economists) further compounds the problem of quality (Pannell 1996 pp. 2-3).

Another related to the difficulty of ensuring that the BCA studies presented with funding proposals were dispassionately rigorous rather than biased by opportunities for scientists to further their own interests at the expense of the public interest ('rent seek') by being strategic with the data and assumptions they provide to BCA analysts. Thus:

> It is too easy to use the BCA framework as 'a method for turning preconceptions into foregone conclusions'. Even if they [scientists] don't cynically manipulate their assumptions, they are well aware that some others do (ibid. p. 3).

A third argument was that BCA at the project level is normally too disaggregated to establish the overall economic accountability of funding decisions by an organisation:

> … [I]t is not theoretically valid to aggregate the benefits of individual projects evaluated separately. Because the BCAs are (and should be) conducted on the basis of changes resulting from a single project, the aggregation of benefits SHOULD add up to more than would occur if ALL of the biological benefits from various projects were to occur. The theoretically correct way to estimate an aggregate benefit from multiple projects is first to aggregate their biological and physical impacts and then estimate a benefit value for these combined impacts. For this reason, BCAs on individual projects should not be used [to ensure accountability in the use of funds, particularly for government-provided funds]. For reasons of consistency, quality and theoretical correctness, analyses at (at least) the program, rather than the project, level, are needed (ibid. p. 3, original emphasis).

The final argument to be discussed here, which follows from the preceding three, is that the results of compulsory project-level BCA studies often lack credibility and thereby undermine BCA analysts' ability to sustain cooperation from those they depend on for data and assumptions:

> [T]o my mind, the most important cost is the considerable damage done to the cause of scientist/economist collaboration. This is always hard to establish, but it suffers greatly when scientists feel that, due to economists, they have had thrust upon them requirements to participate in over-simplistic and rather mechanical analyses, based mainly on guess work (Pannell 1997 p. 3).

These arguments share much in common with the views that Alston et al. (1995) reached from reviewing experiences internationally with agricultural R&D evaluation. They argued that formal BCA methods in this field:

> … are most useful when they are applied at an aggregative, program level. They are less useful at a detailed, project level for at least three reasons. First, the costs of fine-tuning might not justify the benefits in terms of improved allocation of resources to research. Second, measurement problems become increasingly important as the degree of disaggregation increases. Third, micromanaging creative endeavours such as research can be counterproductive; more detailed allocation decisions are probably best guided by well-structured incentive systems instead of

> interventionist, 'hands-on' allocative mechanisms. The last is perhaps the most important conclusion. Formal evaluation and priority-setting procedures should not be used as a basis for replacing ingenuity, serendipity, and scientific entrepreneurship with bureaucratic procedures. There is a wealth of informal evidence that a successful research program rests heavily on the spirit, imagination, judgment, and integrity of agricultural scientists who are allowed freedom of inquiry[11]. The role of research evaluation and priority setting is to help determine the boundaries within which free scientific enquiry occurs (ibid. p. xxii).

It may be argued that differences exist between the R&D and NRM funding spheres that are significant enough to make findings for the former sphere inapplicable to the latter. Confidence in the merits of delegating project-level funding decisions to informal processes involving technical committees and peer review has been justified in the sphere of agricultural R&D by evidence of 'high estimates of historical rates of return [from agricultural research] … [that make it] difficult to argue logically that the seemingly loose *ex ante* evaluation procedures used in the past have not worked well' (Shumway 1981 p. 170). In comparison, substantial concerns have been raised over the efficiency of project-level funding decisions made to date by regional NRM bodies. Nevertheless, those with the concerns have traced much of the problem to perverse incentives arising from poor design and implementation of public funding programs. For instance, Pannell (2001) noted soon after introduction of the National Action Plan for Salinity and Water Quality (one of the two programs constituting the regional delivery model) that the program seemed structured in ways that made it likely regional bodies would spend their allocations of public funds unproductively. Pannell et al. (2008 pp. 1-2) commented further:

> Governments should have provided stronger guidance and support to the regional bodies. …The lack of an agreed decision framework has meant that the focus was likely to be on activities rather than outcomes, and that it would be easy for each organization to avoid responsibility for poor use of public money. … The direction provided by governments was often counter-productive: focusing on on-ground actions through provision of small subsidies to many landholders, rather than well considered and well targeted investments. … [A]s a result of lags in establishing the program, governments emphasized the need to spend the available budget quickly, regardless of the lack of rigorous analysis and planning.

The strongest implication here is that regional delivery arrangements need to be restructured such that perverse incentives for regional bodies to spend funds unproductively are replaced by incentives to target funds more efficiently. BCA, nor any other formalised decision-making procedure for that matter, is unlikely to improve decision-making significantly while regional bodies remain subject to seriously perverse incentives. Consistent with the remarks above, it is very difficult for BCA analysts to guard against regional bodies pursuing their self-interest, as shaped by these perverse incentives, by supplying analysts with data and assumptions favourable to that pursuit.

It might also be argued that ingenuity, entrepreneurship, imagination, integrity and other attributes identified by Alston et al. (1995) as vital for successful R&D are markedly less important for NRM delivery, and consequently their finding that micromanaging the former can be counterproductive is unlikely to apply to the latter. However, the National Natural Resource Management Capacity Building Framework (2002 p. 6) released early in the life of the regional delivery model identified a critical need

[11] Government of Victoria (2007 p. 5-16) identified the loss of such intangible attributes as a cost that should be accounted for in any BCA study of decisions likely to create such loss. With its focus on evaluating regulatory proposals, it remarked that '[a]n example of an intangible cost is the loss of entrepreneurial spirit and behaviour that may arise if the core focus of managers is diverted away from core business activities as a result of regulatory requirements'.

for a decision-making process 'which supports, promotes and encourages innovation, commitment and action'. Publicity surrounding announcement of revised regional delivery arrangements under the new Caring for our Country program signalled that the program would be 'less bureaucratic, reducing the administrative burden on those undertaking activities' (Anon. 2008b p. 1). Moreover, the Business Plan 2009-2010 (Anon. 2008a) for the new program envisages regional bodies exercising considerable entrepreneurship in developing partnerships with other organisations that have complementary skills and interests, including in developing multi-regional and cross-jurisdictional projects.

It would appear from the above that the arguments of Alston et al. (1995) and Pannell (1996, 1997) against routine application of BCA to project-level funding decisions in the agricultural R&D realm do apply in large measure to the realm of NRM funding. A requirement for project-level BCAs should not be imposed on regional NRM bodies as a second-best strategy for mitigating efficiency losses arising from government-created perverse incentives. Besides inefficiencies from not addressing the problem at source, such an imposition of micromanagement would likely result not only in low-credibility funding recommendations but also in substantial losses of the innovation, ingenuity, entrepreneurship and commitment needed from regional bodies. If BCA is to be relied upon in this realm, this would be more appropriate at the level of allocating funds between programs. Decisions made at this level would set the boundaries within which ingenuity, innovation and entrepreneurship are encouraged at lower levels. Consistent with the suggestion of Alston et al. (1995 p. 509) for agricultural R&D, senior management of regional NRM bodies would need to ensure that those delegated responsibility for project-level funding decisions 'are acquainted with the agency's objectives, and the economic arguments related to how they are achieved, in order to ensure that decentralized decisions are made well'.

2.2.9 Limitations of BCA as a guide to allocating limited funds between programs or projects

Six stages of a standard BCA study were discussed in section 2.1. The discussion of stage 5 considered the case where there are multiple discrete options to choose from and the choice is subject to a budget constraint. The 'textbook' procedure recommended there for such a case was to choose the option, or combination of options, that maximises NPV subject to the budget constraint. Presuming the budget constraint applies to all the costs of the various options, this procedure involves calculating a BCR for each option. Identifying the NPV-maximising combination of discrete options begins by ranking the options from highest to lowest according to their BCRs. The analyst moves down the ranked list, including options until the budget constraint becomes binding. The NPV-maximising combination of discrete options contains all options included before the constraint becomes binding. Where the budget constraint applies to all the costs of the various options, this NPV-maximising combination is also that which maximises the aggregate present value of benefits (PVB) subject to the constraint.

Often the range of options available to the decision maker is actually continuous but a limited set of discrete options has been defined to make the analyst's task manageable. This is frequently the case for BCA studies commissioned to identify the optimal allocation of available funds between alternative projects or programs. In the case of allocating funds available to a region between alternative NRM programs (e.g., biodiversity and water programs), for instance, one BCR measure would normally be calculated per program. This measure would be based on a single discrete version of that program in terms of funds invested in it and the portfolio of activities undertaken with those funds.

Alston et al. (1995 p. 371) commented that for BCA of agricultural R&D 'research administrators usually want to know not only *whether* resources should be redirected from program A to program B, they also want to know how much of program A's resources should go to program B – or, indeed, to

program C'. This comment applies equally to the context of regional-scale NRM funding. The BCR for a discrete program is a measure of its *average* PVB per unit of the present value of the available funds invested in it (at least where all program costs are to be covered by those funds). Given the logic of neoclassical welfare economics, however, the PVB-maximising reallocation of funds between two or more programs or projects (which is the NPV-maximising allocation in the context when all program costs are to be covered by the available budget) cannot be identified with data on the average PVB of each program (project). By this logic, identifying this optimal reallocation requires data on the marginal effects on program (project) PVBs of different increments of reallocation (i.e., data on the 'marginal PVB' of each program or project given each level of reallocation). The PVB-maximising allocation of available funds between programs (projects) is that at which the marginal PVB of each program (project) is equal.

Data on the average PVBs of programs (projects), as provided in this context by their respective BCRs, might validly be used in advising decision makers on the *direction/s* in which funds should be reallocated between programs (projects), but it cannot validly be used in advising them on the *degree* to which reallocation should occur. This is because the average PVB of a program (project) is equal to its marginal PVB only in the highly unlikely event of all programs (projects) being characterised by constant returns to scale.

Hence, thorough identification of the NPV-maximising allocation of available funds between alternative programs (projects) requires appreciably more in this context than calculating a single BCR for each program. It requires calculation of multiple BCRs for each program, corresponding to different funding levels for each program (and the activity mix that would be undertaken at each funding level). This increases the workload of the BCA analyst considerably in defining options, gathering the data required to evaluate each, specifying assumptions, and calculating BCRs for each option. Pressures on those the analyst depends on to define options, obtain data and specify assumptions are increased accordingly.

A compromise is therefore normally required between thoroughness and practicality. The compromise suggested by Alston et al. (ibid. p. 371) was to 'consider discrete alternatives … in a way that limits the number considered and sheds light on alternatives not explicitly considered. To do this, we have suggested looking at three alternatives for each program: say 10% above and below a baseline of the current research funding'. Applying this suggestion to a decision problem involving n programs (projects) would require the BCA analyst to calculate BCRs for $5n$ options. The analyst would also need to assess the feasibility (given the overall budget constraint) of the 3^n possible combinations of options. A problem involving four programs would require an analyst following this suggestion to calculate BCRs for 12 options, assess the feasibility of 81 possible combination of options, and compare the aggregate NPV outcomes of the combinations found to be feasible. A problem involving five programs would require calculation of BCRs for 15 options and feasibility assessment of 243 possible option combinations.

Recognising that '[t]he number of alternative allocations to be considered can become very large', Alston et al. (ibid. 372) proposed the solution of 'dealing with a comparatively small number of aggregated programs and by restricting the number of alternatives within each program at the stage of resource allocation decisions'. For instance, a decision problem involving five programs might be simplified by grouping some programs so that the problem analysed involves only three programs. While this makes the analyst's job easier, and might also reduce the pressure on those providing the analyst with data and assumptions, the utility of the analysis to the decision maker is reduced as a result of findings being presented at a more aggregated level.

Even after simplifying the decision problem, the amount of information to be processed by the BCA analyst in approximating the NPV-maximising allocation of funds between programs (projects) will often remain formidable. Alston et al. (ibid.) proposed the use of mathematical programming techniques as a way of the analyst managing this information more efficiently. Although this strategy may increase the efficiency of BCA analysts skilled in mathematical programming, it is typically the case that only a minority of experienced BCA analysts[12] are proficient in this area.

A few final comments are warranted in this section concerning the implications of what has been covered for the costs and other resourcing requirements of estimating the non-market values typically needed for useful BCA studies in the arena of decision making in respect of NRM funding. We saw in section 2.2.4.5 that application of NMV methods tends to be costly, and that this has been a significant obstacle to application of NMV methods in this arena. Part of the cost is monetary (e.g., running surveys). However, substantial opportunity costs normally also arise from the demands of NMV methods on personnel with the requisite sophisticated skills and on the time available for decision making. Each of these resources tends to be in short supply in the arena of NRM funding.

As discussed above, use of BCA to validly inform decisions of how to allocate a given budget between alternative programs (projects) normally requires evaluation of multiple options for each program (project). The non-market environmental effects of each option will typically be unique, meaning that non-market values will need to be estimated for each option. Consider a decision problem involving five programs, where each program generates only one type of non-market environmental effect, and where three alternative funding options (e.g., existing level, existing plus 10 per cent, and existing minus 10 per cent) are defined for each program. Addressing this problem requires 15 (= 5 x 3) non-market environmental values to be estimated. While the requirement to estimate three non-market values per program (project) might not cause a threefold increase in the costs of NMV studies, the costs are likely to be substantially exceed those that would be incurred if only one non-market value were needed for each program (project).

We saw also in section 2.2.4.5 that the benefit-transfer method has been developed as a way of obtaining non-market values when their direct estimation through NMV methods is unaffordable (in terms of demands for funds, time, and/or professional skills). In addition to the issues raised about the benefit-transfer method in that section, the usefulness of applying this method in the context we are concerned with here would depend on its capacity to account validly for how marginal changes in the scale of a program affect its 'production' of non-market values. This requirement would not be satisfied by simply transferring average NMV values (e.g., biodiversity benefit per hectare of remnant vegetation fenced off) from a source study to a target setting.

2.3 Discussion

BCA has been the method traditionally used by economists to evaluate decisions in the public sphere. One of the earliest applications of the method was the U.S. Flood Control Act of 1936 (Chakravarty 1987). BCA studies were performed originally by government agencies – typically by engineers – and usually accepted by the public solely on the basis of agency authority. Questioning of this authority started around 1940, with BCA findings challenged by influential vested interests. This prompted efforts to objectify the procedures of BCA and thereby remove suspicions of bias or incompetence in

[12] The supply of analysts with NRM experience in applying BCA is already small compared with the demand for them that would result if BCA were made compulsory in demonstrating the economic accountability of NRM funding decisions.

its application. Neoclassical welfare economics was largely successful in convincing policy makers and the public that it had delivered such objectification (although, as explained in sections 1.4 and 2.2.3, this apparent achievement relied on various subjective judgements). By the late 1950s, BCA was acknowledged as a respectable specialisation within economics, and it was accepted that the solution to bias or incompetence in its application was to rely on economists to do the job (Marshall 2005; Porter 1995).

In recent decades, however, BCA has come to be used less. Moreover, the findings of those studies carried out, in most instances, wield less influence over decision makers. This seems to be because public policy issues have tended to become more complex and contentious (i.e., 'wicked' as defined in section 2.2.5). BCA was designed originally as a guide for public policy in the context of specific local projects with fairly clear benefits (e.g., whether to invest in flood control or a bridge), whereas public policy, perhaps especially in the environmental realm, has become increasingly concerned with broad-ranging programs seeking benefits that are harder for the public to understand and agree on (e.g., biodiversity conservation) (Gowdy et al. 2007). The declining use and influence of BCA can be attributed also to increasing reliance on other methods of evaluating decisions (e.g., multi-criteria analysis) that have gained credence, including among economists, as more appropriate for wicked problems.

A number of specific reasons were identified in this chapter for the declining use and influence of BCA as conventionally applied by economists. A brief account of these reasons runs as follows:

i. public concerns over assigning monetary values to intangible environmental outcomes (e.g., conservation of biodiversity);

ii public disquiet with subjective judgements of equity implications that are implicit in conventional BCA;

iii controversy over, and costliness of, using non-market valuation (NMV) methods to value unpriced environmental effects of decisions;

iv inability of conventional BCA procedures to accommodate collective value systems emerging from deliberation among parties to a decision;

v conventional BCA analysing problems as if they are mechanistic when they actually behave more like complex adaptive systems;

vi difficulty of accounting for transaction costs in conventional BCA procedures;

vii conventional BCA procedures are often unaffordable (in terms of demands on funding, time and/or professional skills) at the level of disaggregated (e.g., project-level) decisions; and

viii BCA procedures are of limited use by themselves in guiding optimal allocation of funds between programs or projects; they may need to be supplemented for this purpose by mathematical-programming methods, the costs of which further lessen BCA's affordability.

Significant space in the chapter was devoted to reason *iii*. Some of the issues discussed above (e.g., *i* and *ii*) are also relevant to this reason. Briefly, the other issues discussed in relation to non-market valuation methods were:

- disagreement among environmental economists on what neoclassical welfare economics requires from NMV methods;

- questions over the appropriateness of the principle of individual (consumer) sovereignty as a basis for deriving aggregate non-market values;

- lack of opportunity in conventional NMV methods for individuals to deliberate with one another over the information and values against which unpriced environmental effects should be valued; and

- costliness of NMV methods.

An increasing number of economists have reacted to the declining use and influence of BCA in the environmental sphere of public policy, and to the kinds of particular concerns with this method listed above, by either (a) deciding economic evaluation of decisions in this sphere is too hard and/or too unrewarding to justify their continued involvement, or (b) adopting other methods of evaluating decisions that they perceive to be consistent with an economic way of thinking and less compromised by the kinds of problems with BCA identified above.

Nevertheless, BCA remains the default method of evaluation for most economists. Moreover, many governments continue to indicate a preference for this method of economic evaluation, at least for evaluation of regulations proposed by government agencies. For instance, the *Best Practice Regulation Handbook* (Australian Government 2007) states that the 'Australian Government is committed to the use of cost-benefit analysis (CBA) to assess regulatory proposals to encourage better decision making'[13]. Most economists tend to use non-BCA methods of evaluating decisions only when the costs of this method in terms of money, time and demands on professional expertise cannot be justified by the significance of the problem and/or exceed what is available – or when unpriced effects of decisions are important (e.g., allocation of public funds between environmental programs) but valuing these effects by NMV methods (or the benefit-transfer method) is likely to prove controversial. It seems likely that the attitude of most such economists is reflected by Thorpe's (2008 p. 3) observation that 'the use of economics as a tool tends to focus on 'what's doable', rather than 'best practice''.

Some environmental economists have criticised such flexibility within their profession as selling the BCA method short and thus, by their reasoning, undermining the profession's contribution to the quality of public decisions concerning environmental policy. These criticisms seem to be motivated not only by conviction that the methodology of BCA is superior to that of other methods that have found increasing use among environmental and ecological economists. They appear to be motivated also by convictions that democratic political processes left to their own devices are normally prone to reaching decisions far from what economic efficiency would recommend, and that BCA is considerably superior to other methods in providing the discipline on political processes needed to arrive at more efficient outcomes.

Part of the argument here is that 'money talks', so that political processes account much more strongly for those impacts of decisions that are valued monetarily. In the realm of environmental policy, Bennett

[13] However, not all governments are this dogmatic. For instance, the Government of Victoria (2007 p. 5-18) nominated BCA as its preferred method of subjecting regulatory proposals to economic evaluation while recognising that 'multi-criteria analysis is useful when it is not possible to quantify and assign monetary values to all the impacts of an option'.

(2005 pp. 253-254) was concerned accordingly that 'favour may be given to outcomes with readily estimated values. This will predominantly be outcomes that yield marketed, extractive values'. Adamowicz (2004 p. 427) acknowledged that BCA may be methodologically inappropriate for decisions that potentially have irreversible consequences for passive use values (e.g., ecosystem services provided by protected areas), but concluded nonetheless that 'if ignoring passive use values results in placing zero values on such services, then this is likely not a desirable practice'.

The other part of the argument stems from a conviction that political processes tend to be concerned predominantly with *redistributing* existing net social benefits (or 'economic rents') rather than furthering the public interest by *increasing* net social benefits. Economists refer to behaviour designed to capture available economic rents as 'rent-seeking' behaviour. This conviction can be traced to intellectual foundations laid by Buchanan et al. (1962) that spawned various influential economic theories of government, including the theories of rent seeking and public choice. These theories assume all participants in public decision-making processes – including politicians, government officers, special interest groups, experts and scientists – to be self-interested groups just like any other. Along these lines, Olson (1965, 1982) developed an economic theory of collective action that predicted governments would tend to become 'captured' by limited constituencies, since it is easier for these relatively small groupings to act collectively in lobbying governments than it is for the much larger grouping of the general public.

This line of reasoning led to a tendency among policy economists to view themselves as defenders of the public interest against the corrosive pressures of sectional interests (Nelson 1987). It follows from this reasoning that the need for economists to defend the public interest becomes increasingly urgent as political processes become decentralised and thus more under the sway of narrower constituencies. Indeed, most mainstream economists remain unenthusiastic about the current trend to decentralise public decision-making to community-based processes, given that post-WW2 the economics discipline provided arguably the most important intellectual justification for centralised government (Ezrahi 1990; Nelson 1987). Conspicuous in the tendency for economists to promote themselves as defenders of the public interest has been an implicit assertion by them that somehow, unlike other experts or scientists, their agenda (e.g., in promoting BCA and associated NMV methods) is free from the taint of sectional self interest.

One of the main strategies by which the economics profession has sought to defend the public interest from sectional interests has been through seeking to minimise the role of political processes in public decision making. The profession remains influenced strongly by ideas from the 'progressive' movement in the USA that emerged in the late 19th century. This was a time when public confidence in science's ability to deliver progress attained heights that have not been matched since (Nelson 1987). 'Progressives' believed scientific knowledge would steadily expand and gain public acceptance such that political processes would be needed less and less to resolve social divisions and antagonisms. The influence of progressive ideas was particularly strong in the conservation professions from which contemporary environmental management professions descend. Promotion by environmental economists of BCA as a scientific approach to public decision making can thus be understood as motivated by an expectation that wider adoption of this approach would lessen the role of political processes and thus reduce risks of rent-seeking behaviour compromising the public interest. Bromley (2007 p. 677) characterised this as a 'quest for policy without politics … Or, more correctly, it is a quest for public policy in which applied micro-economics is deployed as the only way to impose 'rationality' on an otherwise incoherent and quite un-trustworthy political process'.

This motivation is particularly evident in Bennett's (2005) arguments for why NMV methods should be applied to estimate the monetary value of unpriced environmental effects of public policy decisions. He warned that failures to estimate monetised values of environmental effects mean that 'policy-makers are left to 'judge' the strength of environmental preferences … [and thus] green lobby groups can mobilise their political forces to sway the decision towards environmentally favourable outcomes' (ibid. p. 254). He observed that controversy over the use of NMV methods had increased interest in other decision-making frameworks including multi-criteria analysis (MCA). Observing that MCA requires each of the multiple objectives accounted for in a study to be weighted according to their relative importance, he criticised it because 'the most common way for these weights to be determined [in MCA] is subjectively by either the analyst or the decision-maker, and that can open the door to rent-seeking behaviour' (ibid. p. 251).

Implicit in such arguments favouring increased use of BCA on the basis that this would reduce reliance on political processes is an assumption that the marginal benefits of this reduced reliance, in terms of less 'leakage' of net social welfare due to rent seeking activity, exceed the associated marginal opportunity costs. Part of these opportunity costs involves the additional money, time, expertise, goodwill, entrepreneurial spirit, and so on, that need to be expended (e.g., in applying NMV methods) in order to widen application of BCA and/or obtain monetised values for more of the decision impacts it seeks to account for.

Another part of these opportunity costs involves the benefits foregone as a result of reduced reliance on political processes. While the arguments in foregoing paragraphs assume implicitly that political processes yield only costs, this assumption is not shared in the large and rapidly growing literature that advocates *increased* reliance on deliberative processes (i.e., certain 'small-*p*' types of political process) in environmental decision-making. As discussed in sections 2.2.4.4 and 2.2.5, advocates for deliberative processes of public decision making identify considerable benefits from employing such processes, and argue that such processes are essential in dealing with wicked problems. Indeed, widespread adoption by governments around the world of community-based and other collaborative approaches to environmental management has been motivated by recognition of the advantages often to be gained by devolving such problems to institutional arrangements more conducive to a deliberative mode of politics. To the extent that marginal benefits exist from retaining or increasing reliance on political processes, the logic of neoclassical welfare economics requires that they be traded off against the marginal costs (e.g., from rent seeking) of doing so before concluding that the public interest will continue to be served by supplanting such processes with BCA studies as far as possible.

One leading economist emphasising the benefits of democratic political processes is Bromley (2007 p. 682) who noted that '[o]ne searches in vain for evidence that the democratic working out of local standards of taste and historic commitment has produced outcomes that are Pareto defective … [M]illions of citizens and thousands of local governments have been crafting and re-crafting their villages, towns, and cities for as long as there have been villages, towns and cities. They were doing this long before benefit-cost analysis was created. … And they were doing it long before it was announced that the 'value' of a wetland is what individuals are willing to pay for it not to be destroyed'.

The following remarks from Hajkowicz et al. (p. 45) serve as an appropriate conclusion to this chapter:

> Given the benefits that economics provides, it is easy to conclude that the public and politicians would be better served if economic methods and their application to environmental problems were better understood and more widely used. However, for this to happen economists need to

address the problems and issues in BCA a lot more closely. They need to understand why economics is often ignored as a vehicle for advising decision making, and seek to make economics a more accessible social science. This may mean exploring alternatives in relation to environmental evaluation and conducting BCA.

3. MULTIPLE CRITERIA ANALYSIS

Multiple criteria (or multi-criteria) analysis (MCA) is a generic title given to various methods of evaluating decisions where the performance of options is measured against more than a single criterion. The origins of the MCA approach have been traced to the field of operations research during World War II, where its applications involved military planning (Hajkowicz 2008a). The approach was pioneered following the development of single-objective techniques of mathematical programming, including linear programming. Multi-criteria methods arose out of recognition that many decisions are based on multiple objectives. While the oldest specialist field of MCA is multi-criteria decision making, this field has diversified into various branches including multi-criteria decision aid, multi-criteria decision support, multiple objective decision making, and multi-attribute decision making (Edwards-Jones et al. 2000).

The aim in this chapter is to explain the rationale for, and rudiments of, the MCA approach and consider its advantages and disadvantages as an aid to priority setting in the arena of public funding of environmental programs and projects. The focus is particularly on those settings where responsibility for priority setting resides with community-based, collaborative processes. The rationale for the MCA approach is reviewed in section 3.1, after which the rudiments of the approach are identified in section 3.2. Two particular strands within the MCA approach – cost utility analysis and goal programming – are discussed in section 3.3. Critiques of the MCA approach, both from MCA scholars/analysts and some economists, are considered in section 3.4. Two applications of the MCA approach, each in the context of Australia's regional delivery model for environmental investment, are reviewed in section 3.5. Closing remarks are presented in section 3.6.

3.1 Characteristics and rationale

In simple terms, MCA takes a set of options, a set of objectives to be achieved, and a set of criteria by which performance against these objectives are measured. It evaluates the performance of each option against each criterion. Through a formal procedure it then compares each option's performance against the various criteria to identify the option or combination of options that most successfully satisfies the objectives.

Vatn (2005) observed that MCA methods can be divided into two main classes according to their theoretical foundations. The theoretical foundation of one of these classes is utility based, centred on multi-attribute utility theory (MAUT) (Keeney et al. 1976). Methods comprising this class share with BCA the assumptions that value dimensions are commensurable and that they can be traded-off against each other (compensability). These methods employ aggregating procedures enabling a single value to be computed for each option under consideration. This allows a ranking of the options to be made according to a one-dimensional criterion. Vatn (ibid.) noted that the main difference of this class of methods compared with BCA concerns how the utility implications of options are measured.

The theoretical foundation of the second class of methods assumes that value dimensions are not all commensurable, and recognises accordingly that the potential for trade-offs is limited at least. These methods either avoid aggregation procedures or structure them quite differently from those used by MAUT methods.

As was the case with BCA (see section 2.3), economists did not initiate the MCA approach but have sought to develop and apply it consistent with an economic way of thinking. Nevertheless, BCA has

come to be applied mostly by economists and thus to be regarded as an economic method for decision analysis, while application of MCA continues to be spread over multiple disciplines and indeed often occurs as a multi-disciplinary exercise. Although MCA is no more an approach 'owned' by economists than by any other discipline, it has found widening acceptance among economists as one of the approaches they might apply to better inform decision making. For instance, Gowdy et al. (2005 p. 207) referred to this approach as one of 'the major tenets of ecological economics'.

Hajkowicz (2008a) reported that MCA had mostly received a positive reception among environmental and resource economists, with its main use in these fields as a supplement to BCA when non-market effects of decisions are important. Economists tend to employ MCA when BCA is impractical or unaffordable or when this method is judged ill-suited to the problem at hand. Most economists using MCA have employed MAUT-based methods given their similarity to BCA in assuming different value dimensions are commensurable and can be traded-off against each other. However, some economists (e.g., Vatn 2005) have suggested that non-MAUT-based methods may be more appropriate for some kinds of environmental problems.

MCA was formulated as an approach for decision support in complex problems that typically feature conflict between different interests. As noted by Vatn (2005), existence of conflict implies that interests and values are multi-dimensional and not easily traded-off against each other. MCA was formulated to account for values or criteria that cannot easily be transformed into a monetary measure or any other single metric. Thus it has advantages over BCA when valuation of all effects of decisions in monetary terms is likely to cause controversy that heightens rather than reduces conflict (see section 2.2.2). Black et al. (2004 p. 176) remarked accordingly that '[t]he difficulty and/or objection of converting all benefits and costs to monetary values are the main reasons for the growing interest in multiple criteria tools'.

MCA was developed also to help identify compromises between interests that reduce conflict between them such that solutions can be implemented with affordable political and other transaction costs. Hence, it has advantages over BCA when application of its potential Pareto improvement criterion[14] is likely to 'create a hostile environment for the final decision process' (ibid. p. 337).

The MCA approach is also a response to the bounded rationality of people facing wicked problems (as defined in section 2.2.5). Whereas BCA abstracts from the complexity of such problems so they can be addressed in what seems a comprehensive rational manner, MCA seeks to capture the essential complexity of these problems and serve as 'a structured search process where the analyst supports the decision maker or the actual interest group(s) in defining the problem, looking for alternatives, assessing their consequences, ranking the alternatives, perhaps going back and formulating new alternatives and so on' (ibid. p. 337).

This role is consistent with the call from some ecological economists (Vatn et al. 1994; Norton 2005; Norton et al. 2007) to shift the unit of analysis in environmental decision evaluation from clearly describable solutions to 'development paths'[15]. The goals set in this approach describe favoured development paths rather than involve maximisation of a single metric such as economic efficiency. Hence, the approach moves beyond 'methodological debates about how to force all values into a single

[14] That is, proceed with an option provided 'the gainers gain more than the losers lose' (Vatn 2005 p. 337),

[15] Costanza (2000) proposed a similar approach which involves mutual feedback between a reflective phase of decision making, where social consensus is built around preferred visions of the future, and an action phase, where analytical methods and institutions are applied to help achieve the preferred vision.

measure' (ibid. p. 674). Simulations may be used explore how alternative policy options may lead to distinct scenarios differing in their contributions towards the goals set. The information from simulations of this kind can allow '[p]roposed policies, and the development paths they are modeled to shape and encourage … [to be] evaluated on multiple criteria, including economic criteria (such as job creation and comparative efficiency of different institutional means to achievement improvements on key criteria), but also including longer-term impacts on ecological systems' (ibid. p. 672). These criteria would 'be worked out in the process of building models that are responsive to social problems. This process … ideally includes public involvement as well as agency and managerial participation in an ongoing process that attempts to learn by doing' (ibid. pp. 672-63). The hope is that such social learning 'can 're-model' complex and wicked problems and improve communication by disentangling messes into addressable problems' (ibid. p. 674)

Proctor et al. (2006) observed that MCA studies aim to make explicit the thought processes implicitly undertaken by a decision makers when reaching their decisions. These authors noted that '[i]n complex decisionmaking tasks, which sometimes involve many objectives and many decisionmakers, this structured process may be lost in the complexity of the issues' (ibid. p. 172). Hajkowicz (2007b p. 177) found that '[m]any applications of MCA conclude that its main value is not in providing the 'answer', but from improved transparency, better problem structuring and decision making learning'. Myšiak (2006 p. 273) found from experimental research that 'MCA helps us to gain insight into the decision problem, to consider a larger number of alternatives, and to make better informed decisions'. Described in these ways, MCA shares much in common with the political economy approach to BCA discussed in section 2.2.5. The two approaches aim to foster learning by decision makers by making issues more transparent and facilitating deliberation in the decision process. Unlike the political economy approach to BCA, however, MCA does not seek to value all consequences of decisions in monetary terms.

3.2 Rudiments of MCA

3.2.1 Basic steps

The following basic steps are normally followed in an MCA (Vatn 2005; Hajkowicz et al. 2000; Hajkowicz 2008a):

1. define and structure the problem;

2. define the objectives;

3. define the options (the possible solutions) – a 'without-project' option should be included when this is a realistic alternative[16];

4. define the set of criteria to measure the performance of each option against the various objectives;

5. score each option against each criterion;

[16] Like the without-project scenario for BCA, the without-project option in MCA accounts for the situation when none of the other defined options are implemented. This situation is unlikely to be one in which nothing changes whatsoever. Changes expected to occur in the absence of these options (e.g., farmers' self-initiated behavioural change) should, where relevant to the decision at hand, be accounted for when evaluating the do-nothing option. Proctor et al. (2006) included a do-nothing (called 'business as usual') option in their application of deliberative MCA to subcatchment planning of recreation and tourism in Victoria (see section 3.4.1.1).

6. assign weights to the criteria that reflect their relative values to decision makers;

7. select and apply an algorithm for ranking the options – where relevant, after standardising the criterion scales into units that are commensurable and aggregating the standardised scores across the different criteria;

8. evaluate the result – including by sensitivity analysis – and select the preferred option, or return to step 1 for a second round of the procedure.

Vatn (2005) observed that such a list of steps is not very different from that involved in undertaking BCA, and that the differences are typically found in the way the steps are performed. He noted that the focus on defining the problem is normally more explicit in MCA compared with BCA, and that this follows from the greater acknowledgment in MCA of the complexity or 'wickedness' of the kinds of problems it was developed to address. MCA practitioners normally accept that a problem can be understood in different ways, so that it is important to seek an agreed problem definition before proceeding.

Vatn also remarked that MCA normally proceeds in a more iterative manner than does BCA. The former method's emphasis on learning leaves it more open to returning to earlier steps of the process at any point. This may involve reformulating the options, the objectives and/or the criteria, and in some cases even redefining the problem. Thus '[w]hile the foundation of CBA [i.e., BCA] is the rational agent with given preferences, MCA tries to capture the alternative view that preferences are not clarified. They may rather develop or become clarified as part of the decision making process itself' (ibid. p. 341).

3.2.2 Participation in MCA

As characterised by Vatn (2005), a 'classic' MCA involves a decision maker (e.g., ministry, board of an authority) and an analyst. Although this is similar to BCA, the roles of these parties and how they interact tend to differ considerably between the two approaches. To be consistent with its focus on wicked problems, MCA should strongly involve the decision maker in defining the problem, the options, the objectives, the criteria and the weights. Conduct of BCA, in contrast, is driven much more by the analyst, while the weights given to different value dimensions are obtained by the analyst on the basis of the principle of individual sovereignty (see section 1.4) rather than according to the judgement of the decision maker.

Variations on the classic approach to MCA involving public participation have become more common in recent years. Hajkowicz et al. (2000 p. 48) explained this trend as follows: 'Failure to adequately engage members of the community impacted by the decision may result in overlooking criteria or alternatives of importance'. Vatn (ibid.) noted that variations on the classic MCA approach can involve the participation of stakeholder groups or of citizens, but that the responsibility for the final decision remains with the decision maker. Renn et al. (1993) proposed combining participation of stakeholders and citizens, on the basis that stakeholders may be better able to define what the stakes are (by defining the problem, options, objectives and criteria) while citizens may be better positioned to decide (by setting the weightings) which stakes should be protected.

The classic approach to MCA involves the analyst assuming responsibility for assigning scores to each option against each of the criteria, with the analyst seeking assistance in this task from experts where

appropriate. Applications of MCA driven more from the top down may also involve experts heavily in defining the problem, options, objectives and criteria. Indeed, balancing the participation of experts and the public in MCA studies is often unavoidably complex. In considering this balance, Hajkowicz et al. (2000 p. 52) quoted McAllister (1980 p. 36) as follows: 'Technical experts play an essential role in designing means to specific ends. However, in the selection of ends, the people are the experts by the democratic definition of public welfare. Accordingly, it should never be the role of technical experts to select the ends of public action'.

Spash (2008 p. 277) adopted a somewhat different perspective on this issue, at least in respect of problems as complex as ecosystem management, arguing:

> That experts are making judgments and excluding the public is not necessarily problematic. Instead, we may ask on what ground are appeals being made to the general public in any case? The functions of ecosystems are complex and numerous. Ecologists … effectively frame the issue for everyone else. They know people fail to understand the complexity and importance of the various ecosystem functions. … That information is hard to impart to others merely compounds the problem. … The need is for informed judgment with accountability and this may or may not involve the public. It does however require an open and accessible process.

3.2.3 Defining objectives and criteria

Hajkowicz et al. (2000) described how the objectives and criteria for a MCA study are typically identified through a hierarchical process. The most general objective is stated at the top of the hierarchy, and subordinate objectives ('sub-objectives') contributing towards this major objective are then identified. One or more further rounds of deconstructing the subordinate objectives into increasingly specific sub-objectives may be undertaken before the hierarchy is completed.

Criteria lie at the 'finger tips' of the objectives hierarchy. Following Keeney et al. (1976), Edwards-Jones et al. (2000) identified a number of desirable qualities of such criteria. These are:

- Complete. If two options have the same overall score, then the options must be assessed as having equivalent merit.
- Operational. Each criterion must be capable of being measured in some significant way.
- Decomposable. Achievement of criteria should clearly lead to progress against their overarching (sub) objective.
- Non-redundant. No aspect of the problem should be accounted for more than once, or else the problem of 'double counting' arises which may bias the results in favour of a particular objective.
- Minimal. The set of criteria defined for an objective should be the fewest that can comprehensively account for an objective. This improves efficiency in both data gathering and analysis, and makes the structure of the final analysis clearer.

Hajkowicz et al. (2000 p. 54) emphasised that '[i]dentification of appropriate objectives and criteria in an MCA model is more important than all other considerations such as selection of an appropriate MCA technique … These factors will have the most significant impact on the final ranking of

alternatives as they determine the information inputs to the MCA model'. Especially with MAUT-based MCA methods where weighting of criteria is required, it is important to restrict the number of criteria to that which the decision maker is cognitively able to assign relative weightings to. Bouyssou (1990) was cited as advising that no more than 12 sub-objectives or criteria should be compared by the decision maker at any section of the hierarchy. Achieving this may sometimes entail revision of the hierarchy so that each branch includes a manageable number of criteria or sub-objectives.

Edwards-Jones et al. (2000 p. 155) commented similarly that '[t]he choice of appropriate criteria for measurement in MCA is probably the most critical aspect of any analysis …'. They cautioned that '[t]here are no hard rules governing numbers of types of criteria, but the temptation to focus on easily measurable but relatively unsuitable criteria at the expense of more appropriate ones is an obvious one that should be resisted' (ibid. p. 156).

3.2.4 Standardising the criteria

A major advantage of MCA over BCA is that it can measure the performance of options against multiple objectives, with performance against each objective measured in units relevant for that objective (e.g., dollars, hectares, megalitres). However, MAUT-based methods of MCA require standardisation of criterion measures for each option into commensurate units in order that they can be aggregated into a single measure or rank in terms of the overall utility of each option. Standardisation seeks to eliminate effects of scale that would otherwise introduce a weighting, so that weighting only occurs when weights are introduced explicitly to the analysis (Resource Assessment Commission 1992).

Janssen (1992) suggested two standardisation procedures for criteria where larger scores indicate more desirable outcomes (i.e., greater utility). The first involves dividing each option's score for a particular criterion by the maximum score for that criterion. The second involves assigning each option a score for a particular criterion based on the position of the option's unstandardised score between the minimum and maximum unstandardised scores recorded against that criterion. While various other standardisation procedures have been proposed, none is universally superior to the others. Particular care in standardising scores is required when the scores for some criteria have a much larger range than the scores for some other criteria. The best procedure for each decision problem needs to be determined on a case-by-case basis (Resource Assessment Commission 1992).

For those criteria where it is unrealistic to assume a linear utility function (i.e., that a linear relationship exists between an option's unstandardised score and its utility), it may be necessary to transform each option's unstandardised score using an appropriate utility function before proceeding with the kinds of standardisation procedures mentioned above. For instance, a parabolic utility function may be appropriate where pH is chosen as a criterion of water quality, since the performance of certain ecological functions will decline as pH moves above or below an ideal point (ibid.). A concave positive (e.g., logarithmic) utility function may be appropriate to account for diminishing marginal utility in respect of certain criteria (Hajkowicz et al. 2000; Hajkowicz et al. 2006a). For instance, the utility gained from conserving the first 10,000 hectares of native vegetation is likely to exceed the marginal (i.e., additional) utility gained from conserving the second 10,000 hectares (reflecting the tendency for a given increment in area conserved to be more valuable the scarcer the area already conserved), and the marginal utility from conserving the third 10,000 hectares is likely to be less again. Nevertheless, MCA studies often assume that linear utility functions apply across all criteria, both for computational ease and due to lack of knowledge about the decision maker's actual utility function (ibid.)

3.2.5 Assigning weights to the criteria

MAUT-based methods of MCA require weights to be assigned to each sub-objective and criterion that reflect their relative priority. These weights are normally allocated in a downward-step-wise manner, with weights assigned to higher-order sub-objectives constraining the importance assigned to lower-order (sub) objectives and criteria (ibid.). The weighting procedure involves each sub-objective corresponding with a particular higher-level (sub-) objective, or each criterion corresponding with a particular lowest-level sub-objective, being assigned some fractional degree of importance, with fractional weights adding up to unity (Edwards-Jones et al. 2000).

Weights can be derived in various ways. The Resource Assessment Commission (1992 p. 22) summarised some of the options as follows: 'Weights can be assigned directly by the individual carrying out the analysis to represent hypothetical points of view, they can be based on data obtained from opinion polls, focus groups or other direct forms of sampling public or expert opinion, or they can be generated mathematically from limited information on rankings …'.

Hajkowicz (ibid. p. 57) argued that the weights assigned should reflect the preferences of the decision maker and 'ideally, should be derived through close interaction between the decision makers and decision analyst'. Messner et al. (2006 p. 65) observed similarly that '[i]t might be argued that in a democratic society with elected representatives, it is legitimate that a single person or group of decision makers take a final public decision in the name of the public. Under these conditions, scientists who apply MCA methods must use the subjective preferences of the decision makers in their procedure'. Nevertheless, it is not uncommon for decision makers to be unsure of their priorities (Edwards-Jones et al. 2000), or for decision makers (or indeed any interest group) to be wary about making their true priorities explicit (see section 2.2.5).

Given that the weights should represent social preferences, moreover, the question of who participates in setting the weights can prove controversial. MCA scholars have come increasingly to recognise that decision problems often involve several decision makers as well as multiple other interested parties (Messner 2006). However, their response to this challenge has been constrained by MCA methodology lacking clear guidelines on how to analyse or aggregate multiple weighting schemes.

A common response to this challenge has been to average the different weights, although this means that important information on the extent of preference differences becomes lost (Proctor et al. 2006). Alston et al. (1995) suggested that a Delphi procedure be used in problems with multiple decision makers to arrive at a consensus on weightings for application of MCA to investment decisions in agricultural R&D. In this procedure, each decision maker is asked separately to assign weights to each objective or criteria listed on a form. The answers from all decision makers are averaged, with each decision maker then shown the average weightings as well as their own weightings. The decision makers may meet as a group in the second or third round to justify their weights to one another. The process is repeated until all decision makers are satisfied with the weightings.

Proctor (2001) applied a citizens' jury procedure in seeking consensus on the weights applied in an MCA study of the value of Australia's ecosystem services, and Proctor et al. (2006) used a similar procedure in evaluating options for recreation and tourism in a Victorian sub-catchment. Black et al. (2002, 2004) recommended that this be one approach considered were MCA endorsed for deciding funding priorities under Western Australia's Salinity Investment Framework.

Hanley (2001) proposed that greater objectivity and rigour in the setting of weights would be obtained by identifying them through choice modelling. Black et al. (2004, 2002) identified this as an option for setting weights if an MCA approach were adopted under Western Australia's Salinity Investment Framework. Cleland (2008) applied this approach in exploring how individuals in the Avon Catchment of Western Australia trade off decision criteria against each other. Linares et al. (2002) proposed a goal-programming approach to aggregating preferences of several social groups to obtain a set of consensus weights, and this approach was applied by themselves and also by Marchamalo et al.(2007). Another response has been to move away from the idea of a single 'correct' set of weights for a given problem, towards exploring how different sets of weights (e.g., reflecting the respective priorities of business, national and environmentalist interests) affect the ranking of options (Resource Assessment Commission 1992).

Various methods have been proposed for eliciting information on weights from decision makers or other participants in the decision process. Nijkamp et al. (1990) divided these into two classes: direct methods and indirect methods. Direct methods require the decision maker (and other relevant parties) to state explicitly the relative importance they place on each sub-objective and criterion. A common example is fixed point scoring, where the decision maker is required to allocate a fixed number of points (often percentage points) among the various criteria. A higher allocation of points to a criterion indicates it has greater relative importance (Hajkowicz et al. 2006a). This method is the most direct means of obtaining weights information from the decision maker. However, it can be demanding for the decision maker (and other relevant parties) given that the weighting for one criterion can be increased only by reducing the weightings of one or more other criteria. Indirect methods generally involve the decision maker (and other relevant parties) ranking the options against a set of criteria. Mathematical techniques are then applied to derive implicit weights from the rankings (Hajkowicz et al. 2000).

3.2.6 Ranking the options

A diverse array of algorithms has been developed for ranking decision options on the basis of the information discussed above[17]. Hajkowicz (2008a) observed that some of the more common of these are the Analytical Hierarchy Process, weighted summation, and compromise programming. He observed that research had demonstrated that changing the algorithm can affect the ranking, but that the effect had typically been found to be minor. Arrow et al. (1986) demonstrated mathematically that no ideal algorithm exists, and Myšiak (2006 pp. 273-274) found from experimental research that '[t]here is no single method [i.e., MCA algorithm] which outranks or is outranked by all the other methods with respect to all criteria. Each method has its own strengths and weaknesses'.

The choice of algorithm is dictated to some extent by the nature of the information included in the MCA model and by the nature of the weights. However, choosing the best algorithm for a given problem normally remains a challenge. Aside from technical issues, 'how decision makers would like to interact with the decision model' is often a crucial consideration (Hajkowicz 2008a p. 2). Myšiak (2006 p. 274) remarked that '[s]ome methods [i.e., MCA algorithms] are very complex and it is hard to explain them to scientists or experienced planners who are not used to decision analysis … For this reason, the methods selected are often kept simple and transparent in order to avoid the 'black-box' situation, in which the decision maker feels manipulated and mistrusts the solution recommended'.

[17] Hajkowicz (2008a) remarked that hundreds of algorithms have been developed for this purpose, and referred to Figueira et al. (2005) as providing a recent review of them.

Hajkowicz (2008a p. 2) claimed that '[a]rguably, the most commonly applied MCA technique, possibly by virtue of its relative ease of computation, is linear-weighted summation'. Prato et al. (2007 p. 628) noted too that a utility functions involving linear-weighted summation is frequently used in MCA to rank alternatives '[b]ecause of its simplicity and relevance to real world problems'. This algorithm involves multiplying the standardised criterion scores for an option by their corresponding criterion weights and then summing these products to obtain an aggregate utility score. Aside from the computational advantages of this algorithm, it tends to be easier for decision makers and other interested parties to comprehend. Hajkowicz (2008b p. 609) justified using this algorithm for an MCA exercise where 'stakeholder comprehension and all procedures used to derive a result was mandatory. A complex mathematical procedure that was computationally precise but understood by only a few, if any, stakeholders would not have been accepted'. He further justified his choice on the basis of Janssen's (2001) finding that linear-weighted summation provides a reasonable solution in many applications, and that the most important requirement is to select the right options and criteria.

3.3 A few strands of MCA

The MCA approach comprises a 'broad church' of methods, although some of these are not always recognised as related to MCA. In this section, the links of two such methods – cost utility analysis and goal programming – with MCA are considered briefly.

3.3.1 Cost utility analysis

The method of cost utility analysis (CUA) emerged from healthcare economics in the early 1980s (Drummond et al. 1997), and has found use in the field of environmental and resource economics (e.g., Cullen et al. 2001). Hajkowicz (2008a) noted that the BushTender program in Australia's state of Victoria is based on a similar method.

CUA is useful when the costs of each option are available, but an MCA approach is needed to account for the benefits. The costs are computed by discounted cash flow analysis, as is the case for BCA or cost-effectiveness analysis (CEA).

Whereas CEA is appropriate when the benefits of options can be measured adequately by a single metric (e.g., hectares of vegetation protected), CUA becomes appropriate when the benefits of options can be measured satisfactorily only by using multiple metrics (e.g., hectares of vegetation protected and maintenance of farm business profitability). Performance of options against the different metrics is combined into a single utility metric using MCA. The utility of each option is divided by its cost (in discounted or present value terms), and options are ranked in descending order of their utility per unit of cost (Hajkowicz 2008a). Where options are not mutually exclusive, the results of CUA can be used to identify the set of options that maximises aggregate utility subject to a budget constraint.

3.3.2 Goal programming

Goal programming (GP) represents one strand of the MCA approach (Edwards-Jones et al. 2000; Zografos et al. 2004). GP was developed first in the 1950s (Charnes et al. 1955). In this method, targets are set for the attainment of objectives (or goals). The decision maker can attach weights to each of the objectives reflecting their judgement of the importance of avoiding over- or under- achievement of the respective targets. The GP algorithm then seeks the option or set of options that minimises the weighted sum of unwanted deviations from the various targets (Edwards-Jones et al. 2000).

GP models have been applied widely in analysing natural resource problems. For instance, Walker (1985) used them to analyse reforestation decisions, and Zagrafos et al. (2004) employed them to analyse options for community-based ecotourism. Venn et al. (2007) applied GP in two Australian settings as a way to accommodate indigenous cultural heritage values in natural resource assessment and valuation.

Adamson (2006) proposed applying GP as a 'work-around solution' to the problems of costliness and controversy often associated with efforts by economists to account for non-market environmental values in public decision making. The proposed solution involves not attempting to estimate such values for use in a BCA study but rather accounting for key non-market effects by setting minimum targets that must be achieved in respect of each in order for an option to be deemed feasible. A GP model then identifies the set of options that maximises aggregate net market benefits (the net present value of agricultural productivity benefits in his example) subject to constraints imposed by the available budget as well as the targets.

For Edwards-Jones et al. (2000 p. 160) a key advantage of GP is that it involves 'a logical and easily understood process of analysis, moving from goal definition to achievement in a logical progression'. They described its disadvantages as follows:

> By narrowing down efficiently to a single solution, the final quantity of information supplied to the decision-maker is quite limited; in particular, it does not indicate the trade-offs being made between objectives. In addition, the requirements for information on the preferences of decision-makers are very exacting. In many situations, decision-makers may want decision support to help clarify their own judgements, rather than locate solutions on the basis if predetermined values (ibid. p. 161).

Adamson (2006) recognised also that the task for decision makers in setting targets for non-market environmental effects in a GP model would be challenging in many cases, although he focused on the challenge in terms of the setting of targets leading to 'a pass or fail situation and could theoretically count against a decision maker if the targets were not achieved' (ibid. p. 7). Like Venn et al. (2007), however, he saw GP as a way of reducing the need to subjectively judge how different sub-objectives and criteria should be weighted.

Venn et al. (ibid. p. 335) noted that some economists including Bennett (2005), Bureau of Transport Economics (1999) and URS (2003) had been critical of the subjectivity involved in MCA and had 'assert[ed] that any alternative can be found optimal by varying the weights, the criteria against which performance of policy options are measured and methods for ranking policy options against the criteria'. They argued that the GP variant of MCA 'do[es] not employ arbitrary weights and [is] more appropriately considered as incorporating a mixture of price-based and quantity-based information. Information about the preferences of decision-makers is incorporated in the form of quantity constraints and targets that define the decision space, and weights (shadow prices) that direct the algorithm search for optimal policies within the decision space, rather than unit prices alone'. However, this argument seems to overlook the fact that considerable subjectivity remains in deciding the weights for a GP model and in choosing the levels at which targets should be set for the various objectives.

3.4 Critiques of MCA

A range of critiques of the MCA approach can be found in the literature, including from authors seeing an important role for this approach in appropriate circumstances. Some of the main critiques are reviewed in this section.

3.4.1 Critiques from MCA scholars and analysts

In this section some critiques of MCA from authors sympathetic to this approach are considered. For instance, the Resource Assessment Commission (1992), Hajkowicz et al. (2000) and Edwards-Jones et al. (2000) have each provided lists of the strengths and weaknesses of MCA, and various other authors (e.g., Venn et al. 2007) have discussed the respective merits of BCA and MCA. Table 3.1 presents a synthesis of strengths and weaknesses of MCA as identified in such sources.

Hajkowicz (2008a) acknowledged that MCA is far from a panacea for complex decision problems and identified the following problems often encountered in its application:

- Incorrect problem structure. MCA difficulties can usually be traced to poor problem definition, including in selecting options and defining criteria.

- Poor data on the performance of options (i.e., upon which they can be scored against certain criteria). Sensitivity analysis can help identify the degree to which performance-data uncertainties affect how options are ranked.

- Inappropriate capturing of decision maker preferences. The weighting task is complex and can be misunderstood by decision makers.

- Incorrect application of additive utility. Linear additive algorithms for combining the criterion scores for an option into an overall utility measure, such as the algorithm of linear-weighted summation discussed in section 3.2.6, are often used on the basis that usually they are reasonably realistic. These algorithms assume that the relationship between the overall utility for an option and its score against a particular criterion is unaffected by any of its scores against other criteria. When this is not the case, a multiplicative algorithm may be more appropriate[18].

- Duplicate or overlapping criteria. This problem arises when two or more criteria measure (wholly or partly) the same underlying attribute.

It is not uncommon for critiques of MCA by its sympathisers to discuss the limited influence of results from this approach on the decisions actually made, and conclude that the main value of the approach is not in providing the 'correct answer' but in improving the quality of the decision process (e.g., Ananda et al. 2003; Fernandes et al. 1999; Prato 1999). For instance, the Resource Assessment Commission (1992) reported that:

[18] A multiplicative algorithm treats the criteria as non-compensatory, on the grounds that the most limiting factor should predominate. If an option scores zero on any one of the criteria, its overall score subject to this algorithm will therefore also be zero. Hajkowicz (2006a) noted that a hybrid utility function could be adopted as an alternative, whereby scores for compensatory criteria are added and scores for non-compensatory criteria are multiplied.

Table 3.1: Strengths and weaknesses of multi-criteria analysis

Strengths
Provides structure for decision making while still allowing flexibility.
Particularly useful for complex problems where the amount of information exceeds the integrative capacity of the human brain.
Follows naturally from the way people tend to approach problems with multiple objectives. The effects table constructed in MCA[19] is analogous to a table comparing specifications for several models of car.
Flexible data requirements. Methods are available for qualitative data, quantitative data, or a mixture of both.
Makes the values of the decision explicit through the use of weights and standardisation methods. This helps to make the decision process more transparent.
Allows different points of view to be dealt with explicitly through the use of weights.
Less likely to impose cultural bias on an evaluation (particularly compared with BCA)[20]
Allows information that is agreed on by all parties to be distinguished from areas of contention (indicated by different weights).
Presents results of evaluation in a form which permits interaction between the decision maker and the analyst.
Provides a framework which improves the decision maker's understanding of the decision problem and the trade-offs involved.
Amenable to sensitivity analysis to determine how robust the final results are to changes in the underlying assumptions and methods.
Does not require assignment of a monetary value to all quantities.
Can identify where additional data would be useful and where additional data would have little impact on the final ranking of options.
Can be made very simple or elaborate to suit a particular application and the needs of decision makers. Decision makers can specify the level of complexity with which MCA functions.
Weaknesses
Does not overcome fundamental problems associated with comparing quantities that some would argue are not comparable, but does provide more flexibility than is available with, say, benefit-cost analysis.
Variety of evaluation methods available without any clear indication that one is better than another.
Since many of the methods are complex and remain a 'black box' to the decision maker, they can lead to either mistrust or excessive faith in the results.
Some MCA methods may require an excessive amount of input from the decision maker. Community-based decision makers often do not have sufficient time to engage with such methods.
Concentration on the definition of explicit weights can provide a false sense of objectivity about the remainder of the analysis where weights may have been introduced implicitly through lack of (or inappropriate) cross-criteria standardisation of scores or through the use of inappropriate metrics.
Considerable effort is required to obtain the information for the effects table and the weights.
The impact of weight on the results of MCA are sometimes not understood by decision makers.
Techniques for incorporating time preference into MCA are under-developed.

[19] The effects table is an *m* x *n* matrix constructed for an MCA model comprising *m* criteria and *n* options. It records the weights assigned to each criteria, and the scores for each criterion against each option.

[20] For instance, non-market valuation methods employed for BCA assume respondents are familiar with the purchasing power of money and have satisfactory numeracy skills. These assumptions may not hold in some indigenous societies (Adamowicz et al. 1998). Joubert et al. (1997 p. 139) noted that 'CBA [i.e., BCA] and environmental valuation techniques were primarily developed in the First World. The assumptions inherent in these approaches are less likely to be valid in the Third World, particularly as much of the population is outside of any formal market setting, making the use of CBA questionable'.

> The literature offers no evidence of a decision actually being based on the rankings provided by multi-criteria analysis … This suggests that multi-criteria analysis is merely one input for use by decision makers, and that its usefulness is more likely to be found in its organisational and learning effects and in making explicit the effects of different weightings, than in the straightforward acceptance of its rankings.

Hajkowicz (2007b) studied 55 decision makers who used MCA alongside their own intuitive approaches to decision making under Australia's Natural Heritage Trust program. He found that the conclusions reached by decision makers when MCA was used differed markedly from when they followed an intuitive process[21]. However, the majority of decision makers were unwilling to change their unaided (intuitive) decisions following the use of MCA. Nevertheless, most decision makers tended to agree that MCA improves the decision process through learning, clarification, transparency and accountability, and that this approach should be used to make future investment decisions under the Natural Heritage Trust program.

The overall conclusion from the critiques of MCA sympathisers is encapsulated well by Vatn's (2005 p. 350) finding that:

> While MCA may seem to fit the principal characteristics of environmental decision making better than CBA [i.e., BCA], there are still many problems involved. … While there is obvious potential in MCA, it does not offer simple and unquestionable solutions. The problems should not be seen as a sign of bad methodology. Rather, they indicate various consequences of the kinds of complexities involved not least in environmental decision making.

Hajkowicz (2008a p. 6) concluded similarly that '[a]s with any evaluation tool, MCA has bounded scope for application and introduces methodological challenges of its own. … If sufficient time, effort, and skill are devoted to [addressing the obstacles and errors commonly encountered in MCA], MCA provides a robust and informative evaluation of decision options'. Edwards-Jones et al. (1999 p. 163) focused on the challenge of addressing these obstacles and errors in concluding that '[m]ulticriteria analysis techniques have some clear advantages over more restricted decision-making techniques, such as benefit-cost analysis. … [However] the success of an MCA relies ultimately on the skills of its research team'.

The Resource Assessment Commission (1992 p. 11) found from its review of applications of MCA to environmental decisions that:

> Provided multi-criteria analysis is regarded as an aid to the decision making process rather than a solution to some of the fundamental difficulties involved in resolving resource issues, and provided the technique is applied carefully with due regard to the many subjective decisions that must be made, it appears to have promise as a resource assessment tool and certainly deserves further investigation and application.

[21] He suggested that the reason for this was that MCA may have led decision makers to make more explicit and transparent use of decision criteria.

3.4.1.1 Calls for, and some approaches for paying, greater attention to the deliberative aspect of MCA

Background

An advantage claimed for MCA vis-à-vis conventional BCA, at least in respect of wicked problems, is that it places much more emphasis on facilitating a deliberative decision process which helps decision makers to better understand the problem and thereby develop a MCA model that is more realistic and acceptable to those affected by the decision (see section 3.1). However, some MCA scholars have argued that the approach practised conventionally delivers too little of this advantage. Indeed, Messner et al. (2006 pp. 64-65) claimed that 'the MCA debate does not focus on improving the process of decision making but on upgrading its outcome. … These [MCA] methods are suited to support the competence of decision making by improving information management'. Proctor et al. (2006 p. 173) reported that '[i]n theory and in practice, … MCE [i.e., MCA] does not adequately address the facilitation issue of interaction between analyst and decision makers to elicit and revise preferences …'.

Messner (2006 p. 163) argued also that the procedure in 'early MCA approaches [of identifying] the weights by just asking the decision maker about his or her preferences' was often inadequate, so that 'MCA scientists in recent years have increasingly started to extend their approach to including stakeholder groups and policymakers in the MCA process'. He identified three reasons for this turn towards participatory MCA methods:

- recognition that multiple decision makers are normally involved in any decision process, so that more than one weighting scheme needs to be accommodated;

- acknowledgement that preferences of decision makers are often formed or clarified during a decision process, so that the design of the decision process normally influences the weighting scheme emerging from the process (Roy 1996);

- recognition of the need to increase the opportunities for participation in MCA by affected parties who normally lack political power, and consequently of the benefits for the fairness and legitimacy of decision making from incorporating participatory methods including citizen juries, focus-group techniques, round tables, and the like (Spash 2001).

Paneque Salgado et al. (2009 pp. 2-3) observed similarly that [i]n the last decade multi-criteria evaluation applications … [have evolved] from technocratic to participatory approaches, embedding the context in which decisions are taking place. … The active involvement of interested parties is a necessary condition to legitimise decision-making processes when dealing with complex issues characterised by high stakes and systems' uncertainties'. Meanwhile, there has been an increasing tendency to compensate for the limitations of participatory decision-making approaches by complementing them with MCA methods. Kallis et al. (2006 p. 232) found that:

> … although the procedural benefits of participatory methods are strong, there are important limitations to their instrumental contribution to decisionmaking. … The more participatory processes begin to matter, the more there will be efforts from vested interests to control them. Importantly too, there is a tension between commitment to unrestricted inclusive deliberative participation, and the unavoidable requirement to comply with constitutional, institutional, or policy rules expressing public choice which have been set at different organizational and spatial scales.

Social multi-criteria evaluation

Kallis et al. (ibid. p. 223) applied a variant of participatory MCA termed 'social multi-criteria evaluation' (SMCE), which he characterised as aiming 'to overcome the limitations of participatory processes: it emphasises the cyclic nature of all stages, transparency as an essential component of the evaluation, and reflection and mutual learning between researchers and participants' (ibid. p. 223). The application was concerned with water resources planning in Spain. Sixteen stakeholders – balanced between public authorities, business organisations, and non-governmental organisations – were selected to participate in the SMCE process. These participants were asked through interviews and a follow-up questionnaire to assess the current situation and propose water management alternatives and criteria for evaluating them. Agreement on alternatives and criteria was sought from the participants. The analysts then scored each of the eight agreed alternatives against each of the agreed 11 criteria, using data from specialised literature and technical reports. Overall scores were calculated for the alternatives, and they were ranked accordingly. A focus-group meeting was then held with 11 of the participants to provide them with an opportunity to provide feedback. This process 'proved very important as new alternatives arose …' (ibid. p. 226). Overall, Kallis et al. (ibid. p. 226) found that '[t]he open and participatory discussion process brought to light unconventional judgments of the situation, changing the identification of solutions and the prioritisation of alternatives'.

A number of problems with the SMCE approach were also noted: (i) some powerful actors who had participated in the early stages of the study refused to attend the focus-group meeting; (ii) the approach depended extensively on analysts; and (iii) and the approach lacked deliberation between stakeholders in scoping the problem and in identifying the alternatives and criteria. The dependence on analysts and the lack of inter-stakeholder deliberation may have led stakeholders to 'feel more 'estranged' from the process and its results' than would be the case with a participatory decision-making process without these characteristics (ibid. p. 226).

Participatory integration of MCA with BCA

Messner et al. (2006) reported application of an approach to participatory MCA which integrates BCA with MCA to evaluate large-scale public decisions. The application was concerned with resolution of a water allocation conflict in the Spree River basin of Germany. The approach 'provides a generic framework to structure a participatory evaluation process on public decision issues' (ibid. p. 65). The application involved four steps. The first step of 'problem analysis and scenario derivation' used semi-structured interviews to ask actors in the conflict – including stakeholders and decision makers – to describe their understanding of the conflict and how it might be resolved. This information is used in identifying the set of alternative policy strategies to be evaluated. The second step – 'selection of evaluation indicators and criteria' – involved asking the same actors, in a second round of interviews, to specify the indicators they would like to see applied in evaluating the alternative strategies. The authors explained that:

> In order to prevent disputes among stakeholders and as a matter of fairness, all indicators stated to be important should be included in the assessment process … provided double-counting does not occur and it is feasible to estimate data for them … Later on, evaluation criteria must be defined … for every indicator or group of indicators. … Since the selection of indicators and criteria contains value decisions, it should be done with stakeholders and decision makers (ibid. p. 67).

During the third step of 'impact analysis', participation by the actors 'is limited to the general discussion of models and their assumptions and to data support by experts or stakeholders to adjust models to local conditions' (ibid. p. 67). The fourth step of 'assessment' starts by assessing each policy strategy against each criterion identified in step 2. The results from this initial assessment are explained to a group of participants who:

> ... should be selected in a way that all kinds of interests are represented … [These participants] and decision makers are asked to assign weights to the criteria. Using an outranking approach … rankings of policy strategies are calculated for all participants and these results are subject to discussion. … [T]he aim of the discussion is to find a widely accepted compromise for a weighting scheme … As a result, one or a group of strategies should be identified to be the most advantageous. If it is found that none of the strategies is performing well and some additional alternatives should be considered, an iterative process starts … to take new strategies into account. Proceeding this way, MCA is not used to determine an optimal policy strategy, but to structure the problem and the results, to reveal the uncertainties involved and to feed reliable information as an input into the participatory discourse.

Messner et al. (ibid. p. 73) explained that the application had yet to be completed, but argued that 'even if a common strategy will not be found … the mere application of the … process will have produced a better awareness of future uncertainties and stakeholders [stet] concerns. Thus, the basis of the decision will be much less myopic and narrow than it would have been otherwise'. The downside was that the application was 'neither cheap nor quick', having cost about one million Euro (ibid. p. 73).

Deliberative multi-criteria evaluation

Proctor et al. (2006) applied a new approach for participatory MCA called 'deliberative multicriteria evaluation' to assist a group of natural resource managers choose between recreation and tourism options in the upper catchment of Goulburn-Broken River in Victoria, Australia. The novel approach seeks to combine the advantages of MCA, 'providing structure and integration in complex decision problems, with the advantages of deliberation and stakeholder interaction provided by a 'citizens' jury'' (ibid. p. 169).

The citizens' jury method adapts the model used in Western-style criminal proceedings for application to a public decision-making process. The typical jury ranges from 10-20 participants selected by random, or stratified random, sampling to make it representative of the relevant population. The jury is given a specific 'charge' to deliberate upon. The jury should be given adequate time to deliberate, ask questions, and call 'witnesses' (i.e., experts). A facilitator normally runs the process, which may proceed for several days. The intended final outcome is a consensus position reached by the jury. Proctor et al. (ibid. p. 173) reasoned that '[i]n effect, the citizens' jury approach aggregates multiple preference weights through deliberation to achieve consensus', but that it was usefully complemented by an MCA approach because 'citizens' juries have not addressed the problem of structuring the decisionmaking task' (ibid. p. 173).

The jury for the application comprised six decision makers responsible for choosing recreation and tourism strategies in the case-study region, and therefore was referred to as a 'stakeholder jury' rather than as a 'citizens' jury'. Prior to the 'jury day', a preliminary meeting of jurors was convened to identify a set of management options, objectives for the chosen options, and a set of criteria by which the options could be assessed against the objectives. A small report summarising the meeting outcomes was posted to the jurors for feedback and for their agreement on the precise wording of the options,

objectives and criteria. At the same time a questionnaire was sent to jurors to identify their preliminary rankings of the criteria. The preliminary rankings of some criteria were found to vary widely across the stakeholders. For these criteria, expert witnesses were asked to provide information and respond to questions on the jury day. An effects table, showing the scores of the different options against each of the criteria, was completed with expert assistance from relevant organisations.

The jury was asked on jury day to consider the information in the effects table and presented by expert witnesses, and proceed to reach consensus on a set of weightings for the assessment criteria. The decision processes was assisted by interactive use of a software package that enabled exploration of how different sets of weightings affected the final ranking of options. The morning session of the day involved expert presentations and discussions followed by the jurors providing a weighting (as opposed to just a ranking) of the various criteria. The afternoon session involved iterations of criteria weighting, software interaction, sensitivity analysis, and deliberation. 'Outlier' weightings were identified during this process, and the jurors responsible for them were asked to defend their positions and indicate whether they were prepared to revise their weightings. This process was found to be of crucial importance because of the information it revealed. It allowed jurors to identify the issues of most significance for them, which in turn sometimes led other jurors to reconsider their positions. It was found that 'in general, the stakeholder jury was regarded as a helpful and useful procedure by the decisionmakers, and one which aided them in their understanding of the issues of a complex decisionmaking problem' (ibid. p. 169). However, it was time-consuming. A total of two full days of face-to-face discussion were required from each juror: one day for the preliminary meeting and another for the jury day.

Combining MCA with positional analysis

Positional analysis is a method arising from institutional economics which aims to ''illuminate' an issue or decision situation in a many-sided way to actors and decision-makers who normally differ with respect to how they are positioned in relation to the issue and their ideological orientation' (Söderbaum 2008 p. 113). The method has a ''democracy orientation' [that] goes against the 'technocracy orientation' … where clear-cut solutions and a different kind of expertise are expected' (ibid. p. 114). It employs decision trees as a means of exploring the implications of current and intermediate actions for future options. In neoclassical welfare economics, the focus of using decision trees (e.g., in decision analysis based on expected utility theory) is on the one-dimensional 'pay-offs' (e.g., expected utility scores) at the end of each branch of the tree. In contrast, decision trees constructed in positional analysis are normally open-ended since they refer to an ongoing process. A further difference is that the outcomes of actions are reported not as one-dimensional pay-offs but rather as a 'series of (monetary and non-monetary) outcomes for each period and a series on incompletely known positions that may be understood in qualitative, quantitative and visual terms' (ibid. p. 106).

Söderbaum (2008 pp. 115-116) emphasised the compatibility of positional analysis with participatory approaches to MCA, and suggested indeed that it can 'be regarded as one among the MCAs'. The main advantage of incorporating positional analysis within a participatory approach to MCA would seem to be in accounting systematically for the path-dependency implications of decisions (where the social-ecological systems at issue are complex adaptive systems – see section 2.2.6), and thus for the future cost implications of societal adaptation onto paths that unfolding knowledge reveals to be more sustainable.

3.4.2 Critiques by some economists

Notwithstanding the fact that the MCA approach is finding increased use within the economics profession, including among environmental and resource economists and particularly among ecological economists), a number of negative critiques of this approach have emerged from the profession. Central to such critiques have been arguments that MCA lacks a coherent theoretical basis.

We saw in section 2.2.4.5, for instance, that the leading Australian environmental economist Jeff Bennett (2005) (in his Presidential Address to the 2005 Annual Conference of the Australian Agricultural and Resource Economics Society) referred to use of MCA by economists as an 'avoidance strategy' since this approach departs in some significant ways from neoclassical welfare economics which he held to be the standard for a rigorous economic approach to decision analysis. The (Australian) Bureau of Transport Economics (1999 p. 187) remarked in a similar vein that '[e]xactly what constitutes MCA is hard to say. There appears to be no established theoretical framework or uniform set of principles'.

This Bureau (ibid. p. 193) proceeded to argue that '[b]enefit-cost analysis employs a reasonably well established methodology in specifying and estimating various effects or impacts of a policy proposal on the community. By contrast, the choice of impacts to be evaluated in the [MCA] approach appears to be more arbitrary, because it is not based on an established analytical framework'. They were particularly concerned that MCA lacked the equivalent of BCA's principle of individual (or consumer) sovereignty to provide a systematic guide in choosing which decision effects to account for and in how they should be weighted. They remarked that 'specifying weights is probably the most arbitrary aspect of [MCA] analysis' (ibid. p. 197) and commented further that:

> National interests or community preferences can be reflected in [MCA] by seeking the views of elected representatives, rather than using weights determined by analysts or public servants. However, even governments elected to make judgements about community values and national interests need to be able to defend these decisions for consistency and derivation from some evidence. Also, busy government ministers would be unlikely, at the national or State level, to be able to take decisions on weights used for more than a limited number of projects. Obtaining weights by consulting local communities offers an alternative, but is likely to result in bias towards local interests.

Sugden (2005) stated similarly that '[i]n the judgement of most economists (including the present author), it is a major merit of CBA [i.e., BCA] that it is based on well-understood theoretical foundations, derived from more than a century of research in welfare economics. This gives CBA a high degree of internal consistency'. He noted that it:

> … is particularly significant that CBA has a built-in standard of value: benefits are measured by the maximum amount of money that recipients would pay for them, and disbenefits by the minimum amount of money that recipients would accept as compensation for them. Thus (provided that the assumptions of economic theory hold) the CBA valuation of any given benefit or disbenefit is an absolute amount of money, which the analyst discovers or elicits: it is not defined relative to any particular view about the objectives of the project that creates those benefits or disbenefits. In this sense, CBA does not allow project objectives to be chosen by the government or influenced by stakeholders (ibid. p. 5).

The Bureau of Transport Economics highlighted how a 'consequence of the lack of a framework for choosing impacts in MCA could be double counting. … BCA tends to avoid (but in practice not always successfully …) problems of double counting, because impacts are evaluated as closely as possible to their point of initial incidence' (ibid. pp. 195-196). Sugden (2005 p. 5) noted similarly that:

> The standard of value used in CBA plays an important part in preventing double-counting of benefits and in screening out special pleading. Because benefits are measured by the amount of money that recipients would pay for them, there can be no benefits that are not benefits to specific individuals. This imposes the discipline that a supposedly beneficial project impact cannot be registered in the CBA accounts unless a corresponding class on beneficiaries can be identified, and unless it can be shown that those beneficiaries actually value the impact … It is not open to the government, a project sponsor or stakeholders merely to stipulate that some type of impact is desirable or valuable.

A problem in using MCA to compare options for which a common set of criteria cannot be developed was identified by the Bureau of Transport Economics. They gave the example of trying to use MCA to compare a project to build a road with a project to build a hospital: 'Because the impacts of a road project (travel time, environmental effects, etc.) differ so markedly from the impact of a hospital project (e.g., improvement in health, bed-waiting times, etc.) and are measured in different units – rather than the unifying metric of money values – comparisons would be almost meaningless' (Bureau of Transport Economics 1999 p. 201). Sugden (2005 p. 5) remarked similarly on problems arising in those MCA studies where 'scores are not comparable across projects, only across alternative options for a given project (e.g. different levels of flood protection at a given site). This prevents the score from being used in choosing between projects – one of the main functions of appraisal'.

Sugden was concerned too that adoption of an MCA approach can lead to loss of consistency between evaluations of different projects relative to BCA, even when the sub-objectives, criteria and weightings for each project are taken from the government of the day and applied consistently to each of the projects. He explained: 'Because the cost-benefit studies of different governments and different countries use a common standard of value, a much larger set of studies can be used to test the credibility of findings of any particular one' (ibid. p. 5).

The Bureau of Transport Economics was especially concerned that MCA lacked the theoretical framework of BCA to account for the concept of time preference[22]. They found that '[t]he treatment of time in MCA seems to have been given scant attention in the literature. In particular, it is not clear how criteria that relate to future effects (such as environmental damage) can be summed with effects that refer to the present (such as the net present value of costs and benefits that can be expressed in monetary terms' (Bureau of Transport Economics 1999 p. 200). They did acknowledge, however, that Perry et al. (1978) had proposed the solution of 'time-indexed attributes' in which effects relevant to a criterion occurring in different years would be weighted less the later they occur.

One area in which MCA methods have been used extensively is that of evaluating which agricultural research and development options to invest in. The agricultural economists Alston et al. (1995 p. 463) observed that '[p]erhaps the most common, formalized approach to making decisions on allocating

[22] This concept follows evidence that people tend to prefer a given benefit occurring as soon as possible, and a given cost occurring as late as possible. This concept is handled in BCA through the practice of discounting benefits and costs according to how far into the future they are expected to occur, such that the original benefits and costs are converted into their present-value equivalents (see section 2.1).

research resources is to rank a set of research program objectives according to multiple criteria …'. The scoring methods they referred to in their book are equivalent to MCA methods. They reported that '[m]any scoring studies make little, if any, appeal to a meaningful conceptual framework, a feature that limits their usefulness for any purpose. Often, they also lack a sound methodological basis, and do not proceed logically; they have usually been conceived and executed in an ad hoc fashion' (ibid. p. 465).

Alston et al. (ibid. p. 467) were concerned about the difficulty that decision makers can face in understanding how specific criteria link with their broader objectives: 'While research policy makers may have views about the relative importance of different objectives, they often have little understanding of how the criteria should be combined to generate meaningful measures of the contributions of research to those objectives'. They argued that the theory of neoclassical welfare economics should be used as 'a guide for how to combine these criteria into a useful measure of the contribution of the research to the stated objectives' (ibid. p. 467). More specifically, they argued that criteria relating to total research benefit should selected as appropriate for combining them into an efficiency index corresponding with the concept of economic surplus. They argued that '[o]ne purpose of shortcut methods [such as MCA] is to foster the development of an institutionalized 'economic way of thinking' about research …' (ibid. p. 464).

These authors identified a further limitation of MCA that corresponds with a limitation they identified for BCA that was discussed in section 2.2.9. These limitations arise when either method is employed as a means of evaluating how a given budget should be allocated between alternative programs. They remarked that:

> When approximations to economic surplus relative to research costs have been used [in MCA] to generate priorities, one may be tempted to use the relative size of the efficiency or weighted indexes as cardinal measures of research priorities. In other words, if the net efficiency index for rice [i.e., the rice program as defined by the analysis] is twice as large as the index for wheat [i.e., the wheat program as similarly defined], one might think that rice research should receive twice as many resources – or, perhaps, all of the available resources. This reasoning is incorrect. Particular alternative would have to be scored and ranked if a choice were to be made between them. An example of this would be comparing an existing set of programs with a situation in which there was a 10% increase in crop programs, financed by a proportional reduction in all other programs. Scoring models typically do not include any information about the shape of the research production functions for different research programs. Thus, unlike the mathematical-programming approach to optimization[23], scoring does not allow for any marginal analysis of program changes or optimization within a portfolio of programs. … At best, the ranking derived from a scoring analysis could be used to make the all-or-nothing decision about which programs to support. This could be done by adding up program costs, moving down the ranking, until the total budget for all programs was exhausted – which amounts to treating the results as if they correspond to cardinal rankings, according to NPVs per unit of research resources (ibid. pp. 486-487).

Alston et al. (ibid.) recognised implicitly that it is inappropriate to rush to judgement in criticising the MCA approach for its common practice of varying the weights given to criteria in light of the option rankings they imply. (This practice would seem to be what economist detractors of MCA have in mind when they criticise it – see the closing paragraph of section 3.3.2 – of being a sophisticated means of

[23] This approach was discussed in section 2.2.9, where it was noted that it depends on gaining access to the relatively few analysts with the requisite skills.

justifying pre-ordained decisions.) They observed that even where decision makers have clear views on the relative importance of different criteria, it usually remains difficult for them to assign weights which reflect their value judgements about how criteria should be traded-off against each other. They argued that '[o]ften it is necessary to review the weights in the light of sensitivity analysis that shows the implications of varying the weights, in order to obtain weights that reflect the preferences of policymakers in relation to trading off the various objectives of the research system' (ibid. p. 467).

3.4.2.1 The state of debate

Criticisms of MCA in respect of its theoretical rigour have been taken on board by some MCA scholars. For instance, Hajkowicz et al. (2000 p. 78) found that '[t]he major weaknesses of MCA relate to its lack of rigour and theoretical underpinnings. For example, it does not have the same firm grounding in economic theory as does benefit cost analysis'. More recently, however, such scholars have emphasised more strongly that MCA methods do descend from well-regarded theoretical foundations. For example, Hajkowicz (2008a) observed that the theoretical foundations of MAUT-based MCA methods are found in multi-attribute utility theory developed by Keeney et al. (1976) and in the axioms of utility measurement identified by von Neumann et al. (1944). Other MCA scholars have emphasised the similarities between MAUT-based and BCA approaches, such as how they both assume that performance (utility) of options against different objectives is commensurable and compensable (i.e., trade-offs are possible). Venn et al. (2007 p. 335) observed that MAUT-based MCA is similar to BCA in so far as it is also 'a price-based approach, with practitioner or expert judgment being used in place of economic analysis to derive the relative prices'.

Moreover, MCA scholars have responded to criticisms that it lacks a uniform set of principles by pointing to the desirable qualities of criteria identified by Keeney et al. (1976) (see section 3.2.3). For instance, the desirable quality of non-redundancy among criteria addresses the concern of MCA's economist detractors that the approach is prone to double-counting the effects of decisions. This is not to deny that applying such principles to any problem remains difficult to without an adequate understanding of the problem. Consistent with the position of Alston et al. (1995) as recorded in the previous section, economists have an important role to play in helping decision makers to better understand, by employing an economic way of thinking, the problems they face, and thereby enable them to better define their (sub) objectives and criteria and decide how criteria should be combined to meaningfully measure the options under consideration.

Many economists using MCA remain supportive of neoclassical welfare economics as a theoretical foundation for economic evaluation of public policy decisions, and thus supportive of BCA as the appropriate method of economic evaluation when it is feasible to assign monetary values to all important decision consequences with reasonable accuracy. For instance, Strijker et al. (2000) regarded MCA as a 'next-best solution' to BCA that is nonetheless required when problems arise in validly assigning non-market values to environmental consequences of decisions. Hajkowicz (2008a p. 3) argued similarly that '[i]f benefits are adequately measured in monetary units, then BCA provides an appropriate framework'.

Indeed, a number of economists have integrated BCA with MCA for decision problems where adequate data for applying BCA on its own are not available, or where BCA results for different stakeholder groups are seen as incommensurable. Strijker et al. (2000) identified Nijkamp et al. (1977) and van Pelt et al. (1990) as the source of their idea to apply an 'evaluation method [that] combines the straightforwardness of CBA [i.e., BCA] with the flexibility of MCA. Conceptually, it consists of a MCA, the net result of a CBA being integrated as one of the criteria' (Strijker et al. 2000 p. 363).

Messner et al. (2006 p. 67) applied an integrated approach for multi-criteria decision support wherein BCA results were fed into an MCA, observing that '[a]n advantage of using BCA in the context of MCA refers to the fact that the aggregation of monetised impacts that are incommensurable (e.g. due to equity reasons) need not be done, i.e. the BCA approach may feed several results into the MCA'. Munda (1995) recognised that BCA results could usefully be incorporated in MCA studies, and Prato et al. (2007) included the economic criteria of net return ($/ha) of a farming system and its economic risk ($/ha) among five criteria defined for an MCA study designed to explore how landowners select their preferred farming system.

Sugden (2005) also proposed an approach for integrating BCA within MCA studies as a way of increasing the rigour of the latter. The reasoning for his proposal was as follows:

> Given the merits of CBA as a framework for organising appraisal, it seems highly desirable to retain as much as possible of the structure of CBA within a broader appraisal framework which allows non-monetary impacts to be registered. This can be done in two complementary ways. First, the categories into which project impacts are classified …, representing the 'criteria' or 'objectives' of MCA, can be chosen so that, as far as possible, they correspond with a mutually exclusive and exhaustive classification of costs and benefits that, in principle, are relevant for a CBA. This avoids double-counting and preserves the option of expanding the range of factors that are given monetary values as CBA methodology advances and as data that can be used for benefit transfer accumulate. It also ensures that the monetised entries in the [effects table] are the constituent parts of a limited CBA, i.e. a CBA which takes account only of the monetised impacts. This then makes possible the second way of retaining useful elements of CBA. In addition to the [effects table] which records all impacts, there can be a table which re-displays the monetised impacts as an Analysis of Monetised Costs and Benefits.

Regarding criticisms of MCA for its lack of well-developed procedures to account for time preference, Hajkowicz (2000 p. 74) agreed that a 'significant shortcoming of MCA is its poor ability to handle the temporal dimensions of evaluation. … Although it is possible to include temporal concerns within MCA by adding additional criteria, the theoretical basis for doing so is much weaker. Applications of BCA tend to be much more acutely aware of time than applications of MCA'. When time preference is accounted for in MCA studies, it seems this normally occurs in an unstructured and intuitive manner. The expected temporal pattern of an impact may be reflected implicitly in the score assigned to it, and/or impacts associated with a given criterion may be weighted more heavily the sooner on average they are expected to occur, all else equal. Economists can make a useful contribution in this area in helping to make the process of accounting for time preference in MCA studies more structured and explicit. Even so, economists continue to struggle with the challenge of accounting for time preference when evaluating decisions on sustainability issues which require the interests of future generations to be explicitly considered. The discounting methods they conventionally apply in accounting for time preference account only for the time preferences of current generations. Progress in developing discounting procedures appropriate for sustainability issues remains elusive (Pezzey et al. 2002).

The criticism from Alston et al (1995) that MCA results provide information of limited use in allocating a budget between alternative programs on an efficiency basis (see section 3.4.2) appears to have not been addressed explicitly in the MCA literature. Their concern was particularly over the practice of apportioning an available budget between programs on the basis of their relative overall scores as assessed by MCA (i.e., so that a program with twice the score of another program is allocated twice the share of the budget). They advised that allocation on an efficiency basis requires first that MCA scores be measures of efficiency rather than of equity or any other performance dimension.

Secondly, they argued, it requires ranking programs in descending order of their scores, and fully funding programs in that order until the budget is fully committed.

This criticism and advice is relevant to Australia's regional delivery model for natural resource management, at least in respect of decisions by regional bodies about how to allocate their budgets between their programs and projects. As noted in section 1.3, the language used in policy documents – calling for investments at the regional level to be efficient or cost-effective or to maximise benefits/returns on investment – signals governmental expectations that regional budgets will be allocated among competing priorities primarily on the basis of efficiency.

An equivalent expectation appears not to have been signaled specifically in respect of how the national budget for the regional delivery model is to be allocated between regions. Hajkowicz (2007a p. 209) remarked that '[t]here exist few principles or policy statements on how this allocation [of public funds across regions] should occur', and proposed that the principle of fiscal equalisation (Commonwealth Grants Commission 2004) be employed for this purpose. He regarded this principle as an important indicator of what Australian governments regard as ''fair' as it governs how extremely large financial resources are allocated' (Hajkowicz 2007a p. 210). He interpreted the principle as requiring that 'funds [be] allocated on the basis of need' and applied MCA as a means of equalising the capacity of regional bodies in the state of Queensland to address their NRM needs (ibid. p. 210). MCA was employed specifically to develop an indexed measure of each region's 'relative environmental need, defined by some aggregation of multiple sustainability indicators' (ibid. p. 10). Achieving fiscal equivalence across regions then involved 'the dollar allocation per unit of need, the ratio of funding to need, [to be] the same in each region' (ibid. p. 10).

Nevertheless, the Australian Government (2007 pp. 1-2, original emphasis) states in its *Best Practice Regulation Handbook* that '[t]he challenge for government is to deliver effective and efficient regulation – regulation that is *effective* in addressing an identified problem and *efficient* in terms of maximising the benefits to the community, taking account of the costs. … The policy development process should at least ensure that the benefits to the community of any regulation actually outweigh the costs, and give some assurance that the option chosen will yield the greatest net benefits'. This statement applies to the regional delivery model in so far as it seems to fall within what the *Handbook* defines as '[q]uasi-regulation … which includes a wide range of rules or arrangements where governments influence businesses to comply, but which do not form part of explicit government regulation' (ibid. p. 66). The *Handbook* (ibid. p. 66) seems to be referring to initiatives like the regional delivery model in identifying quasi-regulation as appropriate where 'there are advantages in the government engaging in a collaborative approach with industry, with industry having substantial ownership of the scheme'.

3.5 Use of MCA to maximise returns under the regional delivery model

A number of efforts have been made to use MCA as a guide to how public funds available for investment under Australia's regional delivery model should be allocated between options (e.g., regions or programs) to maximise the return from (i.e., the economic efficiency of) the overall investment. Two such efforts are reviewed in this section.

3.5.1 The assets, threats and solvability model for setting funding priorities in the Wet Tropics region

Hajkowicz et al. (2006a) developed the assets, threats and solvability (ATS) model for MCA in response to 'a need for practically applicable models, guidelines and analytical frameworks that can help decision makers [responsible for environmental priority-setting] resolve trade-offs and direct limited resources towards projects or regions where the expected returns are greatest' (ibid. p. 88). It has been applied in Queensland (Hajkowicz 2002; 2006b) and Western Australia (Government of Western Australia 2003), with its primary role being:

> … to help structure a complex environmental priority-setting problem. It can be used to focus stakeholder debate and provide a standardized means for setting priorities, which makes the process auditable and transparent. The ATS provides a first-tier assessment of priorities which, in some cases, and where resources permit, may be followed up with more detailed appraisals, e.g. benefit cost analysis or environmental/social impact assessment (ibid. p. 88).

Assets in the ATS model are defined as stocks of natural capital which provide flows of benefits to people. The condition of an asset affects the quantity and quality of the benefits it provides, so the ATS model prioritises more highly those assets in a condition yielding more benefits. Threats are defined as 'the degrading processes that cause environmental assets to depreciate in value through time' and which consequently diminish flows of benefits from those assets (ibid. p. 93). The ATS model assigns greater priority to an option addressing a greater threat, all else equal. Solvability is defined as 'the costs involved in addressing the environmental problem under question' (ibid. p. 93). The lower the cost of solving a problem, therefore, the higher its score for solvability. The ATS model places higher priority on more solvable problems 'because there are greater expectations of returns' (ibid. p. 93). It follows from the above that decision options addressing highly valued assets that are under a high level of threat and are highly solvable are prioritised most highly by the ATS model.

Elaboration of the ATS model involves deconstruction of the broad criteria of asset value, threat and solvability into more detailed criteria. This involves a hierarchical process where 'criteria are identified under each sub-branch, with additional nested sub-branches being created where necessary' (ibid. p. 92). The criteria at the 'fingertips' of the hierarchy are those used for evaluating options and can be measured either qualitatively using expert judgements or quantitatively on the basis of datasets. In their application of the model to set funding priorities in the Wet Tropics region of northern Queensland, they used fixed point scoring (see section 3.2.5) to weight the criteria, and linear-weighted summation (see section 3.2.6) as the algorithm for calculating the overall scores of options by which they were ranked. An additive utility function was preferred to a multiplicative one to allow for inter-criterion compensation (e.g., a high score for 'threat' compensating for 'asset value' receiving a low score).

For Hajkowicz et al. (ibid. p. 95) the preferred strategy for allocating available funds between programs or other options (e.g., regions) 'involves ranking the [options] in order of priority and then allocating the scarce resource until a budget constraint binds'. This is the strategy recommended by Alston et al. (1995), as noted in section 3.4.2. However, Hajkowicz et al. (2006a) noted that this strategy is feasible only if the total cost of each option is known[24]. This will often not be the case where the options are defined as broadly as Region A and Region B, or Theme A (e.g., biodiversity) and Theme B (e.g., water). The alternative strategy they proposed (i.e., allocation of funds between options pro rata to their

[24] Black et al. (2002, 2004) observed accordingly that data on the costs of options are needed for MCA to account appropriately for economic objectives.

overall scores) was that which Alston et al. (1995) found to be inconsistent with allocating funds on the basis of efficiency. Hajkowicz et al. (2006a p. 95) observed that this strategy may be more politically acceptable than their preferred strategy since it 'means that each [option] is likely to share at least some portion of the budget', but cautioned that 'proportional funding may not be appropriate where projects have high costs and indivisibility' (ibid. p. 95).

3.5.2 Allocating funds between Catchment Management Authorities in New South Wales

The (New South Wales, NSW) Natural Resources Commission (NRC) was asked to advise the NSW Government on how funds available for investment under the regional delivery model should be allocated between the 13 Catchment Management Authorities responsible for the regions defined for that State under the model. In response, the NRC recommended a six-stage process involving a MAUT-based MCA approach that 'aims to maximise likely return on investment by basing allocations on the priorities of investing government(s) and the likely effectiveness of CMAs in delivering results' (Natural Resources Commission 2008a).

The focus of the NRC's recommended process was on up-front, indicative funding allocations between regions. The NRC (2008a pp. 9-10) described this focus as follows:

> The decision on up-front, indicative funding allocations occurs upstream of other decisions that eventually lead to on-ground investment. … Once the indicative funding allocations are determined CMAs prepare Investment Programs … Investment Programs will outline which parts of the CAPs[25] [Catchment Action Plans] are priorities for investment over the coming 3-4 years and how much it will cost to meet targets and achieve intermediate outcomes. Therefore, the primary purpose of allocating indicative funding to each CMA at the beginning of a program is to ensure that each region will be allocated *broadly* the appropriate proportion of available funding, within which they can confidently plan activities with their communities. These indicative allocations can be thought of as 'forward estimates' that are confirmed annually.

Nevertheless, the NRC (ibid. p. 4) argued that '[t]he government needs to make equally rigorous decisions based on similar principles when approving CMAs' more detailed 4 year Investment Programs to ensure they are strategic and will provide maximum return on investment'. They recommended that the same principles used in defining criteria for assessing indicative funding allocations (i.e., 'priorities' and 'effectiveness' as defined below) be used also in evaluating investment programs, but remarked that 'cost efficiency is another important principle that must be carefully considered when approving Investment Programs' (ibid. p. 36).

Application of BCA to the problem of deciding indicative funding allocations between regions was not pursued because (a) 'there is insufficient data about the likely outcomes from different CMA's investments', (b) 'it is difficult to place a monetary value on NRM outcomes that is comparable across regions and across natural resource themes', and (c) 'it is difficult to transparently incorporate trade-offs between different principles or objectives' (ibid. p. 13). It was reasoned that although BCA had been used elsewhere in setting funding priorities, 'these examples tend to focus on prioritising between *assets*, not between organisations or regions. This is a process we would expect to see undertaken by

[25] Each regional body operating under the regional delivery model (known in NSW as a CMA) was required to develop a regional NRM strategy to be accredited by government before the body could be funded under the model. In NSW, these strategies were called Catchment Action Plans (CAPs) which cover a ten-year period and define the catchment and management targets that the investment planning processes of the CMAs are focused on delivering.

CMAs in developing their investment plans, but is not appropriate for the more high level decision of determining indicative funding allocations between CMAs' (ibid. p. 13, original emphasis).

The NRC (ibid. p. 16) reported that 'NSW and Australian Government agencies indicated that their primary objective is to maximise the likelihood that their CMA-delivered investment will result in improvements in the management and condition of the highest value natural resources. In other words, they are seeking to maximise the likely return on their investment'. They reported further that these agencies had identified two key principles to be considered when allocating funds between CMAs:

'A. Invest in priority natural resource issues (invest where the natural resources are under the greatest threat, in the best condition, where they are most valued by local, state and national communities, and where CMA delivered funding can have the most impact)

B. Invest cost effectively and provide incentives to perform effectively (to generate the greatest improvement for a given amount of funding)' (ibid. p. 16)

The agencies had indicated to the NRC that targeting investment to governments' highest strategic priorities was of primary importance to them. The NRC argued that linking funding to CMA's effectiveness is also important since it 'sets up positive incentives for CMAs to improve their effectiveness and deliver results' (ibid. p. 19).

CMAs identified a third principle that they considered to be as important as the other two principles:

'C. Maintain community momentum (to ensure continuity of community engagement and collaboration, which is a prerequisite for achieving lasting on-ground change)' (ibid. p. 17)

CMAs had argued that their effectiveness in delivering on-ground improvements depended on maintaining their community engagement. However, the NRC did not translate this principle directly into a criterion accounted for in their MCA model. Rather, they used only principles A (called 'priorities') and B (called 'effectiveness') directly. They reasoned that '[t]he need to maintain community momentum is implicit in both of these principles; priorities for CMA-delivered investment will be partially determined by where investment can build on existing community momentum and capacity, and a CMA's effectiveness will depend on the capacity and engagement of their community' (ibid. 17). The NRC (ibid. p. 19) observed elsewhere that its consultation with agencies and CMAs had highlighted a 'divergence of views about how issues of community engagement and capacity should be included in the framework: some believed that community capacity and engagement is a 'means to an end' and should be considered as a component of *effectiveness*; others believed that community capacity to undertake NRM is an outcome in itself and should be assessed as a separate theme under *priorities*'. The MCA model was designed to accommodate either or both of these views.

The NRC (ibid. p. 25) recommended that the criteria for assessing each CMA, or at least its region, in respect of the 'priorities' principle be based on the following propositions:

'1. The highest value regions for CMA-delivered investment are where:

a) the nation's environmental, economic and social values are highly dependent on the landscape functions supported by the natural assets in that region (including where priorities have already been defined in state or national policies eg. Ramsar listed wetlands)

b) natural assets, and hence the landscape functions and values dependent on those assets, are under the greatest threat

c) natural assets that support landscape function are in the best condition compared with the condition needed to support landscape function and values.

2. The greatest potential synergies from CMA-delivered investment can be found where:

d) there is the greatest scope for CMA-delivered investment to add to the regulatory system (for regulating the condition of, or threats to, assets in that theme) and to other players' investments in that theme

e) there is the greatest scope for further CMA-delivered investment to build on the capacity and momentum from past investments in that theme'.

The NRC argued that the starting point for assessing CMAs against the 'priorities' principle should be their respective circumstances in respect of the 13 statewide targets for natural resource management that had been classified into four over-arching themes (biodiversity, water, land, and community)[26], although it emphasised that other priorities (e.g., investment priorities for the Caring for our Country program) could be added to these. It recommended assessing CMAs in respect of the over-arching themes rather than in respect of each of the 13 targets. Assessment in respect of the community theme was 'not assessed in this initial application of the model due to shortness of time and limited available data' (ibid. p. 23).

The assessment criteria associated with the 'effectiveness' principle sought to identify which CMAs are likely to be most effective in delivering government investment. These criteria were based on two readily-available data sources:

'1. CMA plans for investment – measured by the NRC's assessment of confidence that CMA's CAP targets will promote achievement of the state-wide targets

2. CMA progress and results – measured by assessment of progress against the NRC's recommended actions from CAP reviews …' (ibid. 28).

These criteria provide rather indirect indicators of the performance of CMAs in cost-effectively investing the funds allocated to them (as emphasised in the 'effectiveness' principle defined above). Perhaps this limitation explains why the NRC recommended that an additional principle of 'cost efficiency' should be considered when their approach is extended to evaluating CMA's investment programs (see below). The NRC explained that assessing the effectiveness of CMAs in terms of achieving landscape changes targeted in their CAPs was not possible because '[c]omprehensive and comparable data on landscape change as a result of CMA interventions are currently not available'.

The NRC was unable to achieve consensus among stakeholders on the weightings to be given to the various principles and criteria, and thus recommended 'some weights … based on our judgement of the feedback we have heard from stakeholders' (ibid. p. 20). Their weightings were also influenced by the availability of data to assess CMAs against each criterion (e.g., part of the reason for weighting the

[26] These are the targets defined for Priority E4 ('Better Outcomes for Native Vegetation, Biodiversity, Land, Rivers and Coastal Waterways') of the NSW State Plan (NSW Government 2006).

‘priorities’ principle more heavily than the ‘effectiveness’ principle was that in the short term there were more data available to assess the former). Each of the over-arching themes was weighted equally for the purpose of assessing CMAs against the ‘priorities’ principle, although the NRC explained that ‘[g]overnments have the option to indicate their broad preferences for investment in different issues … When deciding whether to weight between themes, governments should consider the biases of the previous programs (lots of funding for salinity, not much funding for coasts) and the potential consequences for regional communities of dramatically shifting priorities in the short term’ (ibid. p. 21).

Each CMA was assigned a qualitative rating for each criterion (e.g., very high, high, low, etc.) and this rating was converted to a numerical score assuming a linear relationship between the possible ratings for a CMA and the likely return from investing investment funds in it. This assumption was made ‘for transparency and clarity, and because of the sometimes high levels of uncertainty associated with each [rating]’ (ibid. p. 29). The ‘raw’ scores for each criterion were then standardised into scores between 0 and 10. The overall score for a CMA, measuring the likely return on investment from allocating funds to it, was calculated by linear-weighted summation.

The overall scores for the CMAs were then used as a basis for recommending the proportions of total available funding to be allocated to each CMA. The proportion of total funds recommended for allocation to a CMA was set equal to its overall score divided by the sum of the overall scores of all CMAs. Thus, ‘if a CMA’s weighted score is equal to 10% of the sum of all weighted scores, then that CMA would receive 10% of the total funds available’ (ibid. 30). Although this approach is inconsistent with the strategy recommended by Alston et al. (1995), as discussed in section 3.4.2, for using MCA results a guide to allocating funds between alternative investment opportunities, and as preferred by Hajkowicz et al. (2006b) when using results from an assets, threats and solvability (ATS) model to guide such decisions, the NRC (2008a) offered no justification for their pro rata approach. The implicit reasons for their choice may have been similar to those offered by Hajkowicz et al. (2006b) for why their preferred approach is sometimes not feasible. Hence, the NRC may have decided against employing the preferred approach because (i) the total cost of program options in each region was not known (given their focus was limited to informing indicative funding allocations between regions), and/or (ii) the pro rata approach was anticipated to be more politically acceptable (since it would ensure that each region receives at least some of the available budget). They noted elsewhere that ‘it is important to undertake a ‘sanity check’ to ensure the outcomes are reasonable and will not lead to unintended consequences or unacceptable environmental, economic and social risks for CMAs or investing governments’ (ibid. p. 33).

3.5.3 Comparing the ATS and NRC assessment criteria

The top-level assessment criteria applied in the ATS and NRC models are listed in Table 3.2 for purposes of comparison. It is evident that the ‘priorities’ criterion of the NRC model encompasses both the ‘asset value’ and ‘threat’ criteria of the ATS model. It also encompasses part of the ATS model’s ‘solvability’ criterion by accounting for the degree to which CMA efforts to solve a threat are made more affordable by initiatives undertaken by others, or by themselves previously. The ‘effectiveness’ criterion of the NRC model accounts for further factors seemingly covered by the ATS model’s ‘solvability’ criterion; namely, factors concerned with how the quality of strategic planning and management by regional NRM bodies affects their cost-effectiveness in solving threats to environmental assets (and thus the affordability of solving these threats). In proposing the ‘solvability’ criterion, Hajkowicz et al. (2006a) presumably had in mind a wider set of factors than covered by the ‘priorities’ and ‘effectiveness’ criteria of the NRC model, which may include:

Table 3.2: Top-level assessment criteria applied in the ATS and NRC models

ATS model top-level criteria	NRC model top-level criteria
Asset value – influenced by asset condition and demand for its services	*Priorities* – influenced by the national value of natural resources, their condition, the degree of threat to them, and the degree of scope for CMA-delivered investment to build on other initiatives (government regulation, non-CMA investments, and previous CMA-delivered investments)
Threat – potential to diminish the value of an asset by degrading its condition	*Effectiveness* – concerned with rewarding CMAs for making progress against performance standards set by the NRC (thereby creating incentives to pursue such progress)
Solvability – affordability of solving the threat	

- the type of threat (e.g., simple or complex);
- the existing (or expected) availability of technical solutions capable of addressing the threat;
- the degree to which solutions are divisible as against 'all or nothing';
- the degree to which those who would need to implement the solutions are willing and able to do so voluntarily (rather than create transaction costs by needing to be coerced, compensated or bribed);
- the degree to which implementation of the solutions is free of political conflict; and
- the degree to which bureaucrats, politicians and others are willing and able to mobilise the resources needed to supplement voluntary implementation and overcome political opposition.

Overall, the top-level criteria defined for the ATS model ('asset value', 'threat' and 'solvability') serve as a more logical and easily grasped starting point, compared with the top-level criteria for the NRC model ('priorities' and 'effectiveness'), for systematically constructing a hierarchy of MCA criteria appropriate for decisions about how regional delivery funds should be allocated to maximise returns from investment. Nevertheless, the NRC approach makes a number of significant contributions in respect of defining sub-criteria for 'solvability'. It makes explicit the need to account for the 'effectiveness' of regional NRM bodies when applying 'solvability' as a top-level criterion in assessing how funds available under the regional delivery model should be allocated between regional bodies when seeking to maximise the returns from investment. It also makes explicit the need to recognise that a regional body's capacity to address a threat affordably depends not only on its current funding but also on capacities that already exist (whether or not the result of the body's prior investments) for it to build on.

3.6 Discussion

Like benefit-cost analysis (BCA), the origins of multiple criteria (or multi-criteria) analysis (MCA) lie outside the economics discipline. Nevertheless, this discipline has been working on applying BCA in accordance with an economic way of thinking for more than half a century, while MCA has begun to attract economists' attention as an alternative to BCA only since the 1990s. As was the case with BCA, economists are steadily exploring ways of applying the MCA consistently with the economic way of

thinking (although not necessarily constrained to the thinking associated with neoclassical welfare economics). In doing so, they have focused largely on MAUT-based methods of MCA which, like BCA, seek to identify the decision option/s that maximise utility.

The primary difference between BCA and MAUT-based methods of MCA is that the former measures the utility of options according to the strict logic of neoclassical welfare economics, whereas the latter permits such measurement according to other logics. In BCA, for instance, the aggregate utility from an option is assumed equal to the sum of utilities that individuals derive from it, and each individual's utility is assumed to be measured appropriately against their personal preferences. MCA is not restricted to these assumptions. The aggregate utility from each option can be calculated, for example, on the basis of preferences (weightings) nominated by the decision-making entity (e.g., board of a regional authority) or as identified by mathematical techniques (e.g., goal programming) or deliberative procedures (e.g., citizens' or stakeholder juries). Moreover, use of MCA enables the multiple effects of decision options to be accounted for more comprehensively in comparing their aggregate utilities. Hence, '[t]he attraction of MCA approaches is that they directly try to address the elements which economists typically mention, but never specify, when referring to 'other factors' as being important in decision processes …' (Spash et al. 2006 p. 386).

As occurred with BCA after World War II, MCA is increasingly being applied by economists, and the respectability within the economics profession of using this approach is steadily growing. This is despite occasional arguments from some adherents to neoclassical welfare economics to the effect that economists using MCA are compromising their professional standards. Hence, MCA is becoming accepted increasingly as an 'economic method' provided it is employed in ways approximating an economic way of thinking – as was the case with BCA more than a half century ago.

Moreover, governments and international bodies are coming increasingly to consider MCA as an alternative or complement to BCA when economic evaluation of a decision is required, or when the economic accountability of a decision needs to be demonstrated. In Australia, the Murray-Darling Basin Commission (1996) recognised MCA to be a cost-effective approach for evaluating projects in the context of integrated catchment management. The *Victorian Guide to Regulation* (Government of Victoria 2007 p. 5-17) listed MCA as one of the tools to be used in determining 'whether a regulatory option is attractive (i.e. its benefits outweigh its costs). These criteria also allow the 'ranking' of different the different options to assess which will yield the greatest net benefit to society'. The volume on project appraisal in the *National Guidelines for Transport System Management in Australia* (Australian Transport Council 2006) includes 'adjusted BCA', a hybrid of BCA and MCA, as an optional technique. Beyond Australia, various international bodies including the World Commission on Dams (WCD) (2000) have acknowledged MCA to be superior to BCA for some decisions requiring economic evaluation. The WCD recommends that MCA be preferred to BCA for decisions regarding investment in large dams in developing countries, for instance, because the former is more transparent and open in dealing with ethical issues such as displacement of indigenous people.

Despite this trend towards greater official acceptance of the MCA approach, this approach still tends to be viewed by Australian governments as 'second best' to BCA, of value when the latter is ill-suited to the particular decision at hand. For instance, the *Victorian Guide to Regulation* (Government of Victoria 2007 p. 5-18) stated that MCA 'is still a second best method of analysis compared to quantitative estimates of costs and benefits, particularly in areas where adequate data should be available'. Indeed, the view that MCA is 'second best' to BCA as an approach for economic evaluation of public decisions remains held by most in the mainstream of the economics profession. To the extent that environmental and resource economists acknowledge a role for MCA in economic evaluation, for

instance, this role is typically seen to be in 'covering' for BCA when a decision involves important non-market effects for which credible monetary values cannot be assigned. For some of these economists, MCA remains at best a 'stop-gap' until problems of validity and affordability in non-market valuation are resolved (e.g., by developments in benefits-transfer techniques).

Such views from economists and governments that MCA is invariably 'second best' to BCA as a method for economic evaluation are inconsistent with the requirement of neoclassical welfare economics that the costs and benefits of an alternative be compared with the costs and benefits of other real alternatives, rather than against some idealised alternative or 'nirvana'. The logical error underlying these views was highlighted by the economist Demsetz (1969 p. 1, original emphasis) as follows: 'The view that now pervades much public policy economics implicitly presents the relevant choice as between an ideal norm and an existing 'imperfect' institutional arrangement. This *nirvana* approach differs considerably from a *comparative institution* approach in which the relevant choice is between alternative real institutional arrangements'[27]. This kind of error has become known as the 'nirvana fallacy'. Avoiding it in the context of this document requires economists not to compare MCA as it is actually practised with BCA as it ideally would be practised (or as it would be practised if certain currently-acknowledged weaknesses were resolved). Rather, it requires comparison of MCA and BCA in respect of how they each would realistically be practised within present circumstances, as well as in terms of how effectively the results of each would actually influence decision making and lead to better decisions.

Moreover, the economics profession concerned with environmental issues has become increasingly fragmented as ideas from the relatively new field of ecological economics steadily gain traction, especially among newer recruits to the profession. Discussions of MCA in this field are more likely to identify advantages for MCA vis-à-vis BCA beyond those of being able to evaluate decisions where credible monetary values of key non-market effects cannot be obtained. These advantages include helping decision makers and interested parties to better understand the problems they face (particularly when the problems are 'wicked', as defined in section 2.2.5, or characterised by increasing-return dynamics that are assumed away in BCA's mechanistic method), and thereby to better define their objectives, their relative preferences for those objectives, and their options for pursuing those objectives. To date, however, most economists advocating use of MCA seem content to rest their case narrowly on its advantages in accounting affordably for non-market effects of decisions. This is perhaps because of fears that broadening their critique of BCA would antagonise their colleagues and thereby reverse gains achieved by arguing the narrow case.

Meanwhile, the balance of the kinds of problems that economists are expected to evaluate appears to be shifting towards those involving complex adaptive systems. This trend is indicated by governments resorting increasingly to polycentric systems of governance[28] (e.g., including collaborative, co-managed, community-based and networked systems) that the literature on complex social-ecological systems[29] (e.g., Walker et al. 2006; Ostrom 1999, 2005; Anderies et al. 2004; Berkes 2007; Marshall 2005, 2008c, 2009, 2008a; Nelson et al. 2008) typically identifies as required for robust governance of

[27] The comparative institution approach is appropriate for choices between alternative methods of economic evaluation since such methods are 'value articulating institutions' (VAIs). A VAI involves a set of rules for the evaluation process. These rules relate to (i) participation (who participates, in which capacity/s, and in what manner), (ii) what counts as data, and (iii) the method by which data is produced and analysed (Vatn 2005).

[28] A polycentric system of governance is one comprising multiple decision-making centres that retain considerable autonomy from one another (Ostrom 1999; Ostrom et al. 1999[1961]).

[29] Anderies et al. (2004 p. 3) defined a social-ecological system as 'an ecological system intricately linked with and affected by one or more social systems'.

such systems. This trend is consistent also with the logic that simpler problems amenable to solution by mechanistic methods will tend to be the ones solved successfully first, leaving complex problems to represent an ever-larger share of the pool of problems remaining to be solved. Where this trend is evident, an accurate matching of method to problem type would be expected to result in increasing reliance on methods like MCA (and/or deliberative methods) better-suited to evaluating wicked and otherwise complex problems, and diminishing reliance on methods like BCA that are better-suited to problems for which mechanistic analysis is appropriate.

The increasing interest in MCA (and deliberative) methods of evaluating decisions is explained also by the decline since the 1970s in the public's trust in expert-driven decision processes. There has been a tendency accordingly for decision processes to rely less on 'black box' methods comprehensible mainly to experts and more on 'glass box' methods more suitable for community-based and other participatory processes. While the BCA method's logic of evaluating options on the basis of their benefits and costs aggregated in monetary terms may initially seem reasonable to much of the public, it is not uncommon for disquiet to arise when some of the 'black box' elements of this method come into view (e.g., underlying value judgements concerning equity, or the reliance on complicated techniques for non-market valuation or 'benefits transfer' of non-market values).

Some of the methods included under the umbrella of the MCA approach also risk alienating the public due to their 'black box' characteristics. However, other MCA methods are suitable for participatory decision processes, and MCA scholars and practitioners have been actively exploring ways of implementing these methods in as much of a 'glass box' manner as possible[30]. Black et al. (2002, 2004) compared BCA, MCA and deliberative methods according to their suitability for participatory decision processes, and ranked deliberative methods the highest against this criterion, followed by MCA methods and then BCA.

The art in employing the MCA approach to support a participatory decision process lies in making the process as transparent, comprehensible and affordable as possible to participants, at the same time as maintaining a level of rigour that satisfies the accountability standards of any governments relied upon to resource implementation of the decisions made. Where demonstrating accountability from an economic perspective is a priority, it is clearly important that the logic by which MCA is applied is broadly consistent with an economic way of thinking.

Although there is no single economic way of thinking, the one favoured conventionally by governments follows the logic of neoclassical welfare economics than underpins BCA. Meanwhile, as noted above, governments seem increasingly aware that contemporary environmental problems are often associated with complex adaptive systems that the logic of neoclassical welfare economics is ill-equipped to analyse. To the extent that this is the case, the economic way of thinking appropriate for demonstrating economic accountability needs to be one consistent with our emerging understanding of complex adaptive systems. This 'complexity' way of thinking[31] includes many of the concepts associated with the neoclassical economic way of thinking (e.g., diminishing returns and trade-offs[32]), while also including other concepts (e.g., increasing returns, multiple equilibria, and resilience) that are foreign to this thinking (Marshall 2005). Whereas BCA cannot incorporate various economic phenomena that complexity thinking identifies as potentially important, the flexibility of the MCA approach enables it, in the hands of a skilled analyst, to take such phenomena duly into account.

[30] Recent Australian examples include Hajkowicz (2007a) and Proctor et al. (2006).
[31] 'Resilience thinking' as popularised by Walker et al. (2006) is a strand of the complexity way of thinking.
[32] For instance, see Janssen et al. (2007).

Regardless of the way of thinking that a government favours as the basis for economic accountability, economists have important roles to play in assisting decision makers, other interested parties, and decision analysts to apply the MCA approach as consistently as possible with the favoured way of thinking, given prevailing limitations of time, data, professional resources, community goodwill, and so on. This role for economists accords with the warning of Black et al. (2004 p. 17) that MCA methods 'must not be seen as a short cut or an easier technique for inexperienced people to use'. Economists can play the role of supporting MCA applications at various levels. As with Hajkowicz et al. (2006a) proposing their assets, threats and solvability model (see section 3.5.1), for instance, economists can develop 'meta frameworks' for MCA that facilitate identification of hierarchies of objectives, sub-objectives and criteria against which the main effects of decisions can be measured consistently with the favoured economic way of thinking. Economists can also work at more fine-grained levels, assisting decision makers and analysts to translate a MCA meta framework into objectives, sub-objectives and criteria that are relevant to a particular decision, including by helping to avoid specification of such metrics in ways that lead to double counting.

Given that the pressures to demonstrate economic accountability under Australia's regional delivery model have increased markedly in recent years (see section 1.3), and the limited use to date of BCA in this area, a number of studies have argued the merits of MCA for this purpose. The resource economists Black et al. (2004 p. 182) stated 'it is in our judgement that multiple criteria tools perform well in NRM contexts' and concluded in a report to the Western Australian Government that MCA-based tools had the greatest potential for satisfying governmental demands for economic accountability under that State's Salinity Investment Framework. In presenting their assets, threats and solvability model for applying MCA to environmental problems, and discussing its application to priority setting under the regional delivery model, Hajkowicz et al. (2006a p. 100) observed that '[t]he aim of the ATS is … to help the decision [for an environmental priority setting problem] withstand subsequent auditing'. Based on a comparison of MCA-based and unaided approaches to decision evaluation under the regional delivery model, Hajkowicz (2007b p. 6) advocated greater reliance on MCA-based approaches because '[t]he practical reality for many environmental decisions is that some form of structured, repeatable and defensible process will be required either under legislation or by auditors'.

Notwithstanding the evident potential of the MCA approach to strengthen economic accountability in environmental decision making, governments would be ill-advised to rely predominantly on this approach for this purpose without adequate safeguards against the risk of decision makers and participants abusing the iterative nature of the process by 'tweaking' a MCA model in ways that rank most highly those decision options best aligned with their sectional interests. As noted in sections 2.3 and 3.3.2, much of the scepticism from economists regarding this approach stems from a belief that any scope for opportunism in the MCA process will normally be exploited through 'rent-seeking' behaviour. Although these 'sceptical economists' tend to regard the iterative nature of MCA as dubious in itself, well-respected mainstream economists including Alston et al. (1995) accept that iteration in specifying an MCA model is often unavoidable if decision makers are to understand the trade-offs in a decision problem well enough to sensibly assign weights to the various decision criteria. Moreover, economists' pessimism that all opportunities for rent-seeking will invariably be exploited may be excessive. Based on her case studies of catchment, landcare and other environmental stewardship groups under the first phase of the Natural Heritage Trust, for instance, Carr (2002 p. 123) observed how:

> Once stewardship groups are successful in securing financial resources and begin implementing projects, they tend to pore over the allocation and disbursal of funds at a level of detail much

more precise than that of normal government accounting systems. Also, groups allocate money, implement projects and make adjustments to expenditure along the way with more alacrity and responsiveness than government budgetary systems. It could be argued that group members have a higher stake in ensuring that their money is well spent than the anonymous government employee has in protecting government coffers.

Nevertheless, prudence requires a government to take reasonable steps in ensuring that any method it endorses for maintaining economic accountability does not become subverted to justify the priorities of sectional interests. This is as true for BCA as it is for MCA, given the ample scope that exists for rent-seeking behaviour to occur in how BCA is applied (see section 2.2.8). In the case of MCA, such steps may involve setting standards of transparency in respect of why the MCA model/s used to justify a particular decision were ultimately preferred over alternative models. Meanwhile, it is important that any such steps taken are indeed reasonable and not heavy-handed. As observed by Schmid (2000), administration of public funding programs in ways offending the dignity of recipient groups runs the risk of provoking them to behave more uncooperatively (e.g., by rent seeking) than they would have otherwise, and thus of increasing the transaction costs of achieving program goals.

4. DELIBERATIVE METHODS

Despite their differences, BCA and MCA are both based primarily on a logic of calculation. In the literature on evaluation of environmental decisions, evaluation methods based on a logic of communication have also received considerable attention. This alternative logic emphasises the role of argumentation or deliberation in defining a problem, and in identifying and choosing between possible solutions. Deliberation was defined in section 2.2.4.4 as involving free public reasoning among equals. Chambers (2003 p. 309) defined deliberation as 'debate and discussion aimed at producing reasonable, well-informed opinions in which participants are willing to revise preferences in light of discussion, new information, and claims made by fellow participants'. For Dryzek (2002 p. 2), deliberation involves communication that promotes 'reflection upon preferences in a non-coercive fashion'. Accordingly, deliberative approaches help people 'reach agreement on the basis of the better argument, on the basis of mutual understanding and trust' (Vatn 2005 p. 350).

The purpose of this chapter is to review the rationale for a deliberative approach to evaluating decisions ('deliberative evaluation'), discuss some of the methods comprising this approach, and consider the advantages and disadvantages of the approach in respect of maintaining economic accountability in a decision process. The rationale for employing a deliberative approach to evaluation is discussed in section 4.1, after which some particular deliberative methods are considered in section 4.2. Issues in the relationship between economics and deliberative methods are reviewed in section 4.3. Some challenges faced in applying deliberative methods successfully are identified in section 4.4, after which closing remarks are presented in section 4.5.

4.1 Characteristics and rationale of deliberative evaluation

Deliberative evaluation is similar to MCA in so far as it can be viewed as a response to the bounded rationality of decision makers and other participants in the decision process. Deliberation in both approaches helps participants to clarify the problem they face and how it should be understood. It helps participants also to better understand each other's needs and perspectives. The key difference arises when in the MCA approach the agreements reached among participants become inputs to a formal model designed to calculate which of the identified possible solutions is/are superior.

The recent interest in deliberative evaluation has its proximate roots in the idea of communicative rationality (Habermas 1984), which has been described as 'a form of common reasoning where consensus is obtained by mutual learning, understanding and changed preferences' (Vatn 2005 p. 351). In her review of the theory of deliberative democracy, Chambers (2003 p. 309) recognised that deliberation does not require participants to forego their own interests, or even to reach consensus, but it does assume they share 'an overarching interest in the legitimacy of outcomes (understood as justification to all affected)'. Dryzek (1990) suggested accordingly that reasoned disagreement can be a more realistic goal for conflict-laden issues. Allowing for reasoned disagreement is consistent with the view that accountability is the conceptual core of legitimacy as interpreted in deliberative democratic theory, where accountability 'is primarily understood in terms of 'giving an account' of something, that is, publicly articulating, explaining, and most importantly justifying public policy' (Chambers 2003 p. 308).

Chambers (ibid. p. 308) characterised the increasing interest in deliberative decision-making processes as a 'turning away from liberal individualist or economic understandings of democracy'. She associated these understandings with 'voting-centric democratic theory' which views democracy as 'the arena in

which fixed preferences and interests compete via fair mechanisms of aggregation'. She referred to deliberative democratic theory as 'talk-centric democratic theory', which 'focuses on the communicative processes of opinion and will-formation that precede voting'. She observed that deliberative democracy is not normally considered as a substitute for representative democracy but rather as an expansion of it.

Advocates of communicative rationality recognise that instrumental rationality, which underpins methods like BCA and MCA focused on finding solutions that maximise utility, should continue to play an important role. However, they are concerned that instrumental rationality has come to dominate spheres of public life where it does not belong. They argue, for instance, that instrumental rationality cannot substitute for democratic political processes in resolving divergent perspectives, differences between values, and conflicts of interest (Stirling 1997). It is argued too that the dominance of instrumental rationality has caused inappropriate commercialisation of public discourse concerning the common good (Vatn 2005). Spash (2008 p. 275) observed accordingly that 'the question of 'consumption for what?' … seems lost as the political economy opts for the simple idea of more consumption as an unquestioned good in itself'.

Proponents of deliberative democracy recognise that reliance on communicative rationality is prone to the risk of strategic manipulation. To the extent that more powerful actors can set the agenda, suppress other's arguments, and limit access to the decision process, they may be able to achieve agreement on their own terms. Two arguments are typically used by proponents of the deliberative approach in answer to threats of this kind (Vatn 2005). The first is that deliberative discourse in itself can constrain strategic behaviour. Since such discourse is about 'we' rather than 'I', parties intending to act strategically must couch their reasoning in ways that are consistent with the common good. This public exposure of their reasoning can serve to constrain these parties from fully realising their original selfish motives. Parties are also constrained in so far as they need to be able to defend themselves plausibly from criticisms that their reasoning is in fact inconsistent with the common good. The second argument is that deliberative processes can be underpinned by institutional arrangements that strengthen their capacity to neutralise vested interests, achieve transparency in agenda setting, strengthen the voice of 'weaker' parties, and so on. Spash (2007 p. 693) acknowledged that 'manipulation is a present and on-going state of affairs in modern society' while arguing that this only heightens the 'need for processes which remove such distortion allowing people to break free and apply some corrective reflection'.

Even when risks of strategic manipulation are manageable, the problem remains that deliberation in itself is not always sufficient to resolve conflicts. Vatn (2005) suggested that the difficulty of this problem is influenced by whether we are dealing with an 'interest conflict' or a 'value conflict'. An interest conflict is one in which the parties involved share a common understanding of the problem and a common set of values, and the conflict is over how the gains and losses due to solutions should be distributed. This kind of conflict may be resolved deliberatively in so far as the force of the better argument may result in consensus over both a preferred technical solution and a preferred distribution pattern for gains and losses (e.g., as modified by compensation paid to losers).

A value conflict is more difficult to resolve since in this case the involved parties lack a common understanding of the problem and a common set of values. In other words, '[t]here is no community across interests' (ibid. p. 354). Environmental conflicts often fall into this category, given that parties to such conflicts frequently find it difficult acknowledging the values of their opponents. Hence, '[t]here is nothing even to bargain about. The interests cannot be reconciled via compensations' (ibid. p. 354). Holland (1997) observed similarly that advocacy of deliberative processes for value conflicts is based on recognition, contrary to conventional economic logic, that parties can persist with non-negotiable

positions for rational reasons (e.g., refusing compensation in order to honour a promise, defend a right, or avoid feelings of guilt and loss of self-identity). Vatn (2005) argued that deliberative processes are needed even more for value conflicts than for interest conflicts, since such processes offer 'the possibility of creating consensus … through changing the understanding and the preferences of those concerned' (ibid. p. 354). Nevertheless, he acknowledged that success in such ventures is far from assured.

4.2 Some methods of deliberative evaluation

Multiple methods of deliberative evaluation have been used (De Marchi et al. 2001). Three of these – focus groups, citizens' juries and consensus conferences – are described briefly below[33].

4.2.1 Focus groups

A focus group comprises small number (10-15) of randomly-selected citizens led by a facilitator. This method seeks to explore the views of the group members on a particular issue in an atmosphere supportive of them openly sharing their ideas, opinions and arguments. The issue to be discussed is defined by the decision maker/s. The discussions are not meant to converge on any recommendation. Instead, a summary of arguments from the focus group is delivered to the decision maker/s for their consideration.

This method offers very limited power to the group members. They are not involved in defining the issue they are to discuss, nor have they any influence over the conclusions that decision makers draw from their arguments.

4.2.2 Citizens' juries

This technique can be applied on a standalone basis or as part of a broader method of evaluating decisions. It is integral to the method of deliberative multi-criteria analysis that was discussed in section 3.4.1.1. As with a focus group, a citizens' jury comprises a small group (10-20 members) of randomly-selected citizens led by a facilitator. Unlike a focus group, however, a citizens' jury is expected to reach a conclusion on the issue put before it and make a recommendation to the decision maker or commissioning body. Further, its deliberations are not self-contained as is the case with a focus group, but supported by 'witnesses' who present additional information to the panel of jurors. The witnesses may be experts or stakeholders. The jury decides what additional information should be presented to it. The process of running a citizens' jury normally runs over 3-5 days. The aim is to arrive at a consensus recommendation. Where consensus cannot be achieved, a voting procedure might be used.

4.2.3 Consensus conferences

A consensus conference shares many of the characteristics of a citizens' jury. It involves a small group of randomly-selected citizens who deliberate on an issue under the supervision of a facilitator. The method originated in Denmark where it has been applied as a way of capturing the perspectives of the public for inclusion in assessments of controversial scientific and technological developments. The citizens comprising the conference can question a panel of experts and stakeholders. They are expected to assess the information before them, discuss the issue among themselves, and arrive at a consensual

[33] These descriptions draw extensively from Vatn (2005).

conclusion. This method emphasises the importance of reaching consensus even more strongly than does the citizens' jury method.

4.3 Deliberative methods and the economics profession

As noted in section 1.4, the mainstream of the economics profession remains faithful to classical liberalism's commitment to protecting individuals' rights of privacy, and thus to methods of evaluating decisions that are based on individuals' independently-formed preferences. This position was bolstered by Arrow's (1951) so-called 'impossibility theorem' which indicates that a stable, unique preference ranking for a group cannot be derived from the preference rankings of its members without infringing on individuals' rights of privacy. The theorem suggests too that any preference ranking agreed on by a group is inevitably influenced by how the decision process is organised, so that any such process is susceptible to strategic manipulation (Arrow et al. 1986). Mainstream economists thus tend to view as undemocratic any effort within a group to agree on a common preference ranking, and argue that democracy is better served by deriving a preference ranking for a group by technical means; i.e., by aggregating the preferences of group members on the basis of the principle of individual sovereignty.

Within the economics profession more generally, a range of leading economists have argued that changes in individuals' preferences arising from argumentation are essential for successful democracy. Buchanan (1954 p. 120) remarked that the 'definition of democracy as 'government by discussion' implies that individual values can and do change in the process of decision making'. Sen (1995 p. 18) observed 'Buchanan is right to emphasize the role of public discussion in the development of preferences (as an important part of democracy)' given that '[m]any of the more exacting problems of the contemporary world – varying from famine prevention to environmental preservation – actually call for value formation through public discussion. Knight (1947 p. 280) found that 'values are established and validated and recognized through discussion, an activity that is at once social, intellectual and creative'. Boulding (1970 p. 118) described as 'absurd' the idea that individuals' preferences should be quarantined from disputation. Randall (1999 p. 32) concluded from his experiences in the environmental policy arena that '[s]tructured discourse and deliberation can often undermine conflict, and careful consideration of information can erode firmly held priors and open up new possibilities'.

Arrow's (1951 p. 7) impossibility theorem followed from his assumption 'that individual values are taken as data and are not capable of being altered by the nature of the decision process itself'. Sen (1995) observed that this assumption is inconsistent with a deliberative conception of democracy which admits the possibility of individuals 'changing their minds' as a result of communicating with one another, such that agreement on a common preference ranking becomes easier. The consensus outside the mainstream of the economics profession on the relevance of the impossibility theorem to a deliberative conception of democracy is summed up well by Spash's (2007 p. 693) statement that '[i]n the light of sustained criticism and empirical evidence the assumptions that preferences are pre-existing, stable and complete across all choice sets, and can therefore merely be called upon, no longer seem tenable'.

4.4 Challenges for deliberative evaluation

The literature on deliberative methods of evaluation has identified various challenges encountered in applying these methods successfully. These challenges are discussed in this section.

4.4.1 Representativeness

The small-group format of the deliberative methods described above makes it easier for participants to establish the mutual trust needed for open communication, and (with citizens' juries and consensus conferences) to arrive at an agreed recommendation. The fewer the number of participants, however, the harder it becomes to represent the variety of interests evident in the relevant public, and thereby to arrive at an outcome (whether summary of arguments or recommendation) accepted as legitimate by the relevant public. Moreover, defining the relevant public is often not clear-cut. In deciding whether to conserve a local patch of bushland, for instance, does the relevant public comprise the local, regional or national population?[34] Once the relevant public is decided, the representativeness achieved from a given sample size can be improved through stratified random sampling, to ensure that the sample covers at least the main categories of the public (e.g., as distinguished by gender, ethnicity, age, occupation, etc.) (Vatn 2005).

One way of assisting outcomes of small-group deliberations to be more representative is to provide group members with information on the views of the relevant public. Opinion polling can be valuable in this regard (Fishkin 1997). Nevertheless, the dominant views as identified by an opinion poll are not informed by deliberative dialogue but rather by respondents' pre-existing understandings.

A related issue concerns whether representatives of stakeholder groups should specifically be included in small-group deliberative formats. Although this may make it harder to achieve mutual trust and open communication within the group, resolution of conflict between stakeholder groups will often not be achieved by excluding these groups from the deliberative process. In any case, stakeholder groups will often lobby to gain access to the process or discredit any agreement that arises therefrom.

Representation of 'silent voices' in a deliberative process is a further challenge often faced in deliberative processes. O'Neill (2001) considered how silent voices can be represented effectively in such processes, and proposed that this challenge might be addressed by finding parties able to speak legitimately on behalf of the silent voices (e.g., future generations, non-human species, disempowered groups unwilling or unable to engage directly with the process). The legitimacy of one party in speaking for another may be judged against multiple criteria including knowledge, expertise, and a track record of actually reflecting the interests of the silent voices (Spash 2007) .

4.4.2 Potential negative aspects of small-group discussions

Black et al. (2002, 2004) noted that deliberative methods are prone to the problems associated with decision making by small groups, notably the problem of discussions becoming dominated by few of the participants. Blamey et al. (2000) prepared a practical guide on the conduct of citizens' juries which suggests strategies for addressing these negative aspects of small-group deliberation.

[34] This challenge is faced also in BCA and MCA. In BCA, for instance, it is faced in defining the population from which a random sample of citizens will be selected to provide data for non-market valuation methods like contingent valuation and choice modelling.

4.4.3 Scope of deliberation

Unless a small-group deliberative process is carefully structured and facilitated, discussion may dwell excessively on preoccupations of the group members. Hence, any conclusions reached may not account adequately for the full range of relevant issues. In considering the appropriateness of the citizens' jury method for Western Australia's Salinity Investment Framework, for instance, Black et al. (2004 p. 178) observed that this method 'can potentially overlook the costs of treatments and outcomes as there is less structure in terms of information required for the jury to make a decision upon. Efficient use of funds may be completely overlooked in favour of say, ease of implementation'.

One solution to this challenge involves the decision maker/s imposing some structure on the deliberative process. An alternative is to employ deliberative methods of evaluation in combination with other methods that provide the necessary structure, as occurs with the political economy approach to BCA (see section 2.2.5), deliberative monetary valuation (see section 2.2.4.4) and deliberative multi-criteria analysis (see section 3.4.1.1). Although the political economy approach to BCA was envisaged originally as a dialogue between the analyst and the decision maker/s, this dialogue could be broadened to include a wider range of interested parties. However, this approach does require all key effects of decision options to be valued in monetary terms. Even where this is feasible, it will not always be acceptable to all participants.

Other methods that seek to add structure to a deliberative process of evaluation include social multi-criteria analysis (see section 3.4.1.1), scenario analysis and mediated modelling (Kallis et al. 2006), and positional analysis (Söderbaum 2008). Application of scenario analysis typically involves a preparatory phase followed by a two-day workshop. In the preparatory phase, organisers send stakeholder-participants four contrasting scenarios about the future of the issue under question. This is followed by a vision-making phase on the first day of the workshop, during which participants employ the scenarios to articulate, deliberate on, and agree on a future vision statement. The final phase of idea generation and action planning involves the participants identifying the measures required to help realise their agreed vision. Possible actions are debated, voted on, and ranked. The result is an action plan that details the measures to be implemented, allocates responsibilities for implementation, sets a timetable, and identifies the resourcing requirements (Kallis et al. 2006).

Mediated modeling aims to develop 'scoping-level' models suited to understanding dynamic behaviour patterns rather than predicting outcomes in a precise manner. These models are developed interactively with participants. Qualitative models are first developed which depict the main variables describing the system. This model is then quantified using information provided by participants. The computer-based quantified model is employed to analyse the dynamic consequences of different management scenarios. A mediated modeling exercise typically involves two to four full-day workshops. The number of participants can range from few (5-12) to many (50-100) (Kallis et al. 2006).

As indicated in section 3.4.1.1, positional analysis utilises decision trees as a basis for exploring an issue in a pluralistic manner. It is particularly useful for social-ecological problems involving complex adaptive systems, since it systematically examines the implications of a particular decision for path dependency and consequently for the costs of societal adaptation as new knowledge unfolds.

4.4.4 Limited understanding and interest of participants

A prerequisite of deliberative evaluation methods is that participants have sufficient interest in, and basic understanding of, the issue under discussion that they can contribute meaningfully to the process.

As noted in section 2.2.4.3, environmental issues are often of a complexity that non-experts find difficult to comprehend, leading them too often to base their preferences on 'warm and fuzzy' feelings. Unless the issue is warm and fuzzy, or attracts their interest in some other way (e.g., through media coverage due to controversial aspects of the issue), moreover, it may prove difficult to engage a representative sample of citizens in a deliberative process due to lack of interest. Black et al. (2002, 2004) concluded accordingly that deliberative methods tend to perform better when the issue to be discussed is topical and focused. They observed that issues in natural resource management are often broad-ranging rather than focused (reflected by the emphasis on integrated approaches to addressing issues in this field) and that this can make deliberative methods less appropriate in this domain.

4.4.5 Lack of trust from participants

Spash (2007 p. 694) remarked on the importance of prospective participants trusting those who urge them to participate in a deliberative process, and observed that '[w]here responsible agencies fail to follow through o the recommendations of consultation future attempts to consult will likely prove unpopular and levels of distrust will be increased'. Trust may also be an issue for stakeholders used to being given priority in decision-making processes. Such stakeholders may baulk at joining deliberative processes if they lack trust that their arguments will carry as much weight as they do in conventional decision-making processes, or at least that the outcomes will be as favourable to their interests as they are accustomed to.

4.4.6 Problems in striving for consensus

The citizens' jury and consensus conference methods of deliberative evaluation each strive for consensus among the participants. Spash (2007 p. 692) acknowledged that methods seeking consensus have a role to play, but cautioned that 'the emphasis on consensus politics has also been overplayed and a 'consensus' approach may be used to silence minority opinions rather than empower'. He observed that '[c]onsensus can be achieved more easily where issues are generalized, but once specific plans of action and detail arise divisions tend to appear' (ibid. p. 692). While the possibility of 'win-win' solutions is often held out as a carrot for people to participate in deliberative exercises, such solutions often lack credibility because obstacles to their adoption, or their potential side-effects, have been overlooked. It may be preferable in some cases not to persist in seeking consensus but rather to facilitate the expression of dissenting voices.

4.4.7 Disincentives for governments to sponsor deliberative processes

A disincentive for governments to sponsor deliberative processes is that '[o]nce people have been engaged in a process of deliberation and been empowered as part of a decision process their presence and opinion is less easily dismissed' (Spash 2007 p. 694). An unpalatable finding from a BCA process may be less constraining on a government because 'a price which is too high or too low can more easily be adjusted (in private), subject to sensitivity analysis (using expert manipulation), dismissed (via judicious use of alternative expert opinion), or if necessary recalculated (hiring another consultant and informing them of the 'failures' of the last study)' (ibid. p. 694).

A further disincentive to governments using deliberative methods in a process of economic evaluation is that many politicians and public officials do not know of any decision evaluation methods other than BCA (Söderbaum 2008). Aside from ignorance of deliberative methods due to lack of exposure to them during their formal education and subsequent career, these methods often do not correspond with what politicians and officials were taught to expect from a method of policy analysis. Thus, 'specific

expectations on scientists and experts have been institutionalized over the years. An expert advising politicians should come up with clear-cut solutions, it is believed. The analyst should be able to rank the alternatives and be able to identify the 'best' option from a societal point of view' (ibid. p. 116). As systematic and rigorous as deliberative evaluation may be, the fact that it offers conditional rather than precise conclusions often remains an obstacle to governmental acceptance of it as a method of policy evaluation.

Aside from politicians and officials having become acculturated to favour methods delivering them precise conclusions, they also have a vested interest in preferring methods delivering them clear-cut answers to controversial problems. Unequivocal answers from analysts make it 'possible for politicians to hide behind this expert judgement. Responsibility is shared with the expert or in the extreme case, shifted to the expert' (ibid. p. 116).

4.5 Discussion

Within academic and some policy circles, the interest shown in deliberative methods of evaluating decisions has increased markedly over the last decade. Chambers (2003 p. 307) reported that '[i]t is now commonplace to talk about the deliberative turn in democratic theory', and 'this turn is so striking that it has spawned a small industry of review articles and edited volumes attempting to sum up its meaning and content'. This trend has been evident in the domain of environmental governance, with Neef (2009 p. 54) reporting that 'the first years of the 21st century have seen a growing convergence between the natural resource literature on public participation and deliberative democratic theory'. This convergence is reflected in policy circles where there has been increased focus on collaborative models of environmental governance in place of the prior reliance on 'episodic and symbolic forms of democratic participation' (ibid. p. 54).

According to Black et al. (2002, 2004), moreover, the available evidence (e.g., Rippe et al. 1999; Nancarrow et al. 2002) indicates that the public tends to prefer deliberative processes such as citizens' juries to benefit-cost analysis. In particular, Black et al. (2002, 2004) referred to a survey by Nancarrow et al. (2002) of 260 respondents in the Moore catchment to the north of Perth in Western Australia. Between 60-80 per cent of respondents agreed with each of the following statements: 'You can't really decide NRM priorities by analysing the costs and benefits in dollars', 'If the decision making process is fair, people should accept the NRM funding decisions', and 'All sections of the community have a right to have a say on setting priorities for NRM funding'.

Nevertheless, a tension normally exists between the theoretical ideal for deliberative processes and the 'unavoidable requirement to comply with constitutional, institutional, or policy rules expressing public choice which have been set at different organisational and spatial scales' (Kallis et al. 2006 p. 232). Such constraints imposed on deliberative processes may 'jeopardise the very idea of engagement and participation'. One such constraint involves the expectation that a decision process will be transparent such that it can feasibly be reconstructed and, if queried in the political process, justified (Messner et al. 2006). Proctor et al. (2006) indicated that this expectation is often not met in deliberative processes because much of the logical thought processes that occur remain implicit. Part of the problem here is that deliberative processes tend to be used for complex problems, where the line of logic followed often gets lost in the complexity of the issues. Proctor et al. (ibid.) suggested that MCA be used in tandem with deliberative methods as a framework for breaking down a complex problem into workable units, as well as structuring the problem, so that complexities can be disentangled and it becomes easier to make transparent the reasoning behind the decision finally reached.

Meanwhile, formidable challenges remain in creating and maintaining the conditions needed for participatory processes to run deliberatively, rather than adversarially as most citizens have come to expect from political processes. Kallis et al. (2006) noted that many applications of deliberative methods have been exploratory and linked only loosely to actual policy processes, and that it remains unclear whether the benefits observed from such applications would arise in applications more likely to influence policy decisions. They found '[t]here are good reasons to suspect that the greater decision impact of a participatory process, the more that the process would be governed by power games, strategic behaviour, and attempts by powerful vested interests to capture the process' (ibid. p. 232). Lane's (2003) analysis of the Regional Forest Agreement process in Australia offers a good example of how partisan dynamics can come to dominate decision-making processes launched by governments using the language of deliberative democracy.

The complexity of many environmental issues presents further difficulties for deliberative participation of the public in this policy domain. The knowledge needed to engage meaningfully with complex issues may be confined to experts, and it may be costly and time-consuming to bring others 'up to speed' so they may also participate effectively. Limiting deliberation in such circumstances to the pool of relevant experts may be justified. However, it is necessary on such occasions to guard against experts pursuing their narrow strategic interests, or else unconsciously applying their own subjective judgements when questions of value arise (Refsgaard 2006). Hence, deliberative processes dominated by experts need to be transparent and accountable to the broader public (Spash 2008). MCA has been proposed as a way of augmenting expert-driven deliberative processes to make them more transparent to laypeople (Refsgaard 2006).

Deliberative approaches have also been criticised in some quarters for avoiding hard decisions. For instance, Black et al. (2002, 2004) observed that community participation in NRM funding decisions in Australia has led too often to most issues and programs being rated a high priority.

It seems unlikely, therefore, that deliberative methods alone will be sufficient for meeting the demands for greater economic accountability when using collaborative, community-based approaches for deciding how public funds should be allocated between competing environmental priorities. Nevertheless, the advantages of such approaches for promoting constructive deliberation were central to the rationale for adopting them. Methods of establishing economic accountability need therefore to incorporate a strong deliberative dimension if they are to be consistent with this rationale. As shown in chapters two and three, scope exists to incorporate deliberative methods within both BCA and MCA, although MCA lends itself more readily to this accommodation.

5. INVESTMENT FRAMEWORK FOR ENVIRONMENTAL RESOURCES (INFFER)

Some Australian economists have not been content to engage only in academic debates over the most appropriate methods for strengthening economic accountability under community-based approaches to environmental decision making, and have chosen also to work directly with policy makers in helping them respond constructively to the strengthening calls for economic accountability under such approaches. Although the interest of Australian economists in this role can be traced to the early 1990s (e.g., Marshall et al. 1993), their actual involvement in this area remained limited until David Pannell and other economists associated with the School of Agricultural and Resource Economics at the University of Western Australia began from 2000 onwards to argue in favour of an economic framework for prioritising public investments in addressing salinisation problems in that state.

Although numerous tools, models and frameworks have been developed to support environmental investment decisions, INFFER was singled out for review in this document because of its explicit focus on being adoptable within the context of community-based management, and because it has achieved significant levels of endorsement and use by governments and community-based environmental bodies in Australia.

An account of how INFFER evolved from its origins in Western Australia's Salinity Investment Framework (SIF), and of how it has come to be applied, is presented in this chapter. The purpose here is partly to explore the political economy surrounding the development and use of economic frameworks in this area, as a guide to the kinds of issues that may be encountered in the present project. The purpose in this chapter is also to consider INFFER in light of the discussion in the preceding chapters. The evolution of INFFER from the SIF is discussed in section 5.1. Comments on various elements of the method used in INFFER are presented in section 5.2. Closing remarks are provided in section 5.3.

5.1 Origins and evolution of INFFER

5.1.1 Background to the Western Australian Government's Salinity Investment Framework

Dryland salinity is a serious natural resource degradation issue in the south-western agricultural zone of Western Australia (WA). In 1996, the WA Government launched a Salinity Action Plan in response to this issue. This strategy emphasised the need for an asset-based approach to prioritising investments. It established the concept of 'recovery catchments' within which priority would be given to restoration/protection of: key water resources; high-value natural diversity and wetlands; and designated infrastructure. It also established the concept of 'focus catchments' to describe catchments where agricultural values were high or a catchment committee was prepared to take action. This emphasis on prioritisation reflected recognition by the WA Government that the effectiveness of agency support services in the past had been limited by spreading them too thinly (Cleland 2008)[35].

The Salinity Action Plan was replaced by the State Salinity Strategy (Government of Western Australia 2000) in 2000. Although the new strategy placed greater emphasis on participatory planning, capacity building and cross-sectoral partnerships, it maintained an asset-based approach to prioritising

[35] This reference was drawn on extensively in preparing sections 5.1.1 to 5.1.8.

investments. Nevertheless, it elaborated this approach by distinguishing catchments according to whether they would be managed by the principles of 'recovery', 'containment' or 'adaptation'.

5.1.2 Establishment of the SIF

Presentations to the State Salinity Council by David Pannell and others led the chair of the Council to propose the formation of a steering committee to develop a State Salinity Investment Framework. The starting point for developing such a framework was a paper by Pannell et al. (2000), the rationale of which was endorsed at the first meeting of the working committee. A set of principles based on this rationale was subsequently endorsed by the Taskforce as the appropriate basis for future investment decisions at a state and regional level. These principles became known as the SIF (Salinity Investment Framework). The principles were officially endorsed by the WA Government in 2002 as a statement of policy. These principles are presented in Table 5.1.

Table 5.1: Salinity investment principles documented in a policy statement released by the Western Australian Government on 12 March 2002[36]

1. *The top priority public investments are those which generate the greatest public benefits per dollar of public investment*. Whether protection of a particular asset falls into the 'top category' depends on the costs of preventative treatments, the effectiveness of the treatments and the values of the assets. 'Values' includes social and environmental values, as well as economic values.
2. Direct financial assistance to landholders to undertake salinity action should be strategic and should not exceed the public benefits that result. It should be focused on priority areas with high value and high probability of success.
3. *Where the priority is high and net public benefits are sufficient, Government should be prepared to take strong action to ensure protection of the asset.* This might include compensation or structural adjustment, regulation and monitoring to ensure achievement.
4. *Where the public priority is low but there are extensive private assets at risk, public investment should be aimed at industry development.* Appropriate industry development is a activity that seeks to develop profitable systems to prevent or contain salinity or to adapt to saline land and water.
5. *Inevitably, a targeted investment strategy in salinity management will result in an unequal distribution of investment across the State.* Government must accept, and the community appreciate, that the limited funds available for public investment can only target assets where there is the highest public benefit.
6. *Government must fulfil its statutory obligations for land, natural resources and functions (such as research) when it sets its priorities for investment in salinity action.* These obligations are required by law and should be taken into account early in the planning process. This emphasizes the need for people to better understand socio-political processes at all levels.
7. *The process required for priority setting will involve ongoing learning and need constant feedback.* Over time, funding priorities will change as new information becomes available and programs adapt, goals are met and new challenges arise. Feedback will be assisted by making assumptions explicit at all stages, and especially so when assessing and re-assessing feasibility of options.
8. *Setting priorities must proceed where there is only limited or imperfect information on prevailing environmental, social and economic circumstances*. Doing nothing because not everything is known is generally not a good response. The impact of and lessons learnt from early actions should be used to improve understanding of catchment processes.

5.1.3 The role of community in applying the SIF: rhetoric versus reality

Nevertheless, some concerns were expressed that application of the investment principles would lead to a top-down process of priority setting. The chair of the South West Catchments Council wrote to the

[36] This table reproduces Box 1.1 in Cleland (2008).

State Salinity Council arguing that regional NRM groups 'need to be involved in determining the use of public funds for the management of Regional issues' and that 'every region will differ in striking a balance between economic investment and the need to attract maximum community involvement'. A comment from another source was that '[a] bureaucratic, academically driven funding allocation process would disempower the community and delay useful movement on NRM in WA by a decade' (Cleland 2008 pp. 158, 159).

WA's Avon NRM region was chosen for a trial application of the SIF, which commenced in the second quarter of 2002. The trial was to occur both at the state level and at the regional level. The project brief for the trial envisaged the state-level and regional-level applications being developed in parallel, with priorities set at one level influencing priorities at the other level. However, Cleland (ibid. p. 218) found that 'a more hierarchical relationship existed with regards to the administration and day-to-day management of the two applications' and that this kind of relationship 'filtered down' to the community consultation component of the trial. She observed that this led to perceptions that 'the SIF was a purely top-down directive and prompted questions about the role of bureaucrats/experts; the scientific legitimacy of the state-wide application; and the authenticity of the consultation process' (ibid. 218).

A particular reason for SIF becoming driven from the top down related to its perceived value in helping agencies respond to the pressures to have regional NRM strategies and investment plans developed and accredited in accordance with national guidelines defined for the two main programs comprising the 'regional delivery model' introduced by the Australian Government early in the new millennium: the National Action Plan for Salinity and Water Quality (NAP) and the second phase of the Natural Heritage Trust (NHT2). Cleland (ibid. p. 274) noted that '[e]arly on in the specification phase of the SIF it was recognised that the SIF process needed to be linked to the national accreditation process and that the results of prioritisation needed to be tested in regional strategies'.

5.1.4 Developing an asset-based approach

The Working Group distinguished four classes of assets of relevance to the SIF: land, water, biodiversity, and infrastructure. These asset classes corresponded closely with the existing responsibilities of NRM-related government agencies in WA. The Working Group discussed whether the SIF should be used to allocate investment funds between these asset classes. Although most members agreed that the SIF had an important role to play in cross-class allocation, a compromise was reached of initially allocating within the four classes and then subsequently considering allocations across the classes. Cleland (2008 p. 187) remarked that '[t]his decision was in line with traditional administrative structures which established clear management and funding silos for specific NRM resources'.

The involved government agencies each developed a method for setting priorities within their respective asset classes. The community workshops held in October 2002, where the agencies were to explain their own methods, provided impetus for moving towards greater consistency. However, 'a SIF update issued in November 2002 noted that progress on making the agency processes more consistent was 20 per cent complete and that this was a generous score' (ibid. p. 203). It was decided by early 2003 that 'value' and threat' would be the common criteria to be used initially for priority setting within each class. An approach was developed whereby agencies would arrange relevant assets in a value versus threat matrix. However, a report released later that year found that, despite this common approach, agencies differed in how they were measuring threats (Department of Environment 2003). The report noted also that neither the agency methods nor the 'threat-value matrix' considered

prioritisation across asset classes. They also did not account for multiple benefits where different assets were situated near one another.

5.1.5 Social assets

The goals of the State Salinity Strategy were concerned with 'providing communities with the capacity to address salinity issues and to manage the changes brought about by salinity' (Government of Western Australia 2000 p. 10). Subsequent SIF documents clarified that assets of public value 'could include any element of water, land, nature, infrastructure or community, whether publicly or privately owned or managed, which provides a public service or benefit' (Cleland 2008 p. 267).

From these considerations the term 'social assets' arose. However, confusion arose regarding this term's meaning and its relevance for the SIF. Arguments arose over whether social assets are means to biophysical ends or whether they are ends in themselves. David Pannell argued strongly against 'investing NRM funds in social structures and networks just for the sake of having these things' (ibid. p. 268).

Nevertheless, community workshops for the SIF trial in the Avon region allocated significant time to identifying the region's social assets. A conclusion from the workshops was that it was necessary to include social assets in the SIF as a distinct asset class, and a sub-committee of the SIF Steering Committee agreed to develop a series of social asset types for use in the state-level trial of the SIF. The consultancy resulting from this undertaking argued that the social assets listed in the consultancy report (summarised as appendix 5 to the SIF1 final report (Department of Environment 2003)) needed to be considered in relation to their relevance for natural resource management. The types of social assets listed by the consultancy, and their constituent items, are presented in Table 5.2.

A draft project brief for SIF2 recognised that 'the quality and quantity of social assets in the geographic vicinity of an asset can significantly affect community capability to manage salinity', and envisaged a key output of SIF2 as being a list of priorities for community capacities (Cleland 2008 p. 269). Early development of the SIF2 methodology considered social assets 'in terms of (1) an assessment of the socio-political will/capacity to apply adequate resources to achieve the goal for the [biophysical] asset and (2) whether the option would be implemented or supported by surrounding land managers' (ibid. 270).

5.1.6 Evaluation of methods for operationalising the SIF

Disagreement within the SIF Steering Committee on the question of whether the SIF should be used for prioritising investments across asset classes meant that the question remained unresolved. One view was that 'prioritising across assets was the actual intent of the SIF' (Cleland 2008 p. 207). The other view, argued by one member in particular, was it is 'not possible to compare apples and oranges' such as by comparing the merits of investing in a particular wetland versus a particular road (ibid. p. 206). In an attempt to resolve the issue, the Steering Committee commissioned a consultancy to explore, among other questions, whether it were possible to develop a valid method for prioritising investments across asset classes. The consultants were economists from the School of Agricultural and Resource Economics at the University of Western Australia.

The consultancy report (Black et al. 2002) found that the SIF had established decision rules without providing an analytical framework by which these rules might be applied in practice. It evaluated the potential of three types of decision support method – BCA, MCA and deliberative methods – to fill this

Table 5.2 Social asset types and items identified in the SIF1 final report

Asset type	Asset items
Knowledge and skills	Knowledge and skills available Ability to grow knowledge and skills Robustness and availability
Values / culture	NRM values Sense of place, cultural heritage Robustness, persistence, resilience and availability
Community well-being	Community health Cohesiveness
Networks / organisations	NRM values Quality of social interaction Information flow Learning capacity
Economic resources	Investment available from businesses reliant on natural resources Investment available from sources not reliant on natural resources
Governance capacity	Governance capacity

Source: Department of Environment (2003 p. 125)

gap. The overall conclusion was that that an MCA tool should be applied to the SIF for prioritising investments. The particular recommendations included the following specific comments:

- ‘Multiple criteria tools provide a sound framework for complex decision-making problems. They are particularly suited to NRM because the tool: (1) distinguishes a broad set of criteria used in NRM decisions; (2) can effectively identify trade-offs between conflicting objectives, and (3) deal adequately with non-monetary, qualitative and uncertain information’.

- ‘The main criticism leveled at multiple criteria tools is the arbitrary nature of their weighting system. A logical progression in overcoming the problem is to incorporate either economic or deliberative techniques to generate the weights’.

- ‘Clear guidelines should be prepared for the practical application of the multiple criteria tool to the Salinity Investment Framework’ (ibid. p. 46).

The consultancy report also suggested a greater focus be placed in the SIF on accounting for multiple benefits arising from investing in particular assets, to ensure that its application was not biased in favour of investments protecting single assets. Also highlighted was the challenge of integration across funding programs in order to maximise the benefits from the total pool of funds available for enhancing environmental quality.

The member of the Steering Committee who had been particularly opposed to the idea of SIF being used to prioritise investments across asset classes argued that the consultancy report ‘did not convincingly support the view that there are cost-effective means between different asset classes’, and proposed instead that the decision on allocating funds between asset classes be left to politicians

(Cleland 2008 p. 210). Michael Burton, one of the authors of the consultancy report, and David Pannell argued in response that formal analyses of allocation decisions between asset classes were required to make the trade-offs explicit in order that politicians could make informed choices. David Pannell motioned to support the report's recommendations, arguing 'that the multiple criteria approach was systematic, transparent and participatory', and all members other than the dissenting member 'voiced that they agreed in principle to the recommendations but no vote was taken on the matter' (ibid. p. 210).

Although it was then recommended that experts review whether government agencies' priority-setting methods were compatible with the MCA approach, '[t]his directive dropped off the agenda …' (ibid. p. 210). Reviewing this experience, Cleland (ibid. p. 207) formulated the proposition that 'an individual on the [steering] committee had a strong agenda to refute the logic of prioritisation across assets, which prevailed throughout the consultancy and the SIF project at large'.

5.1.7 A second phase for the SIF

The initial timeframe for trialling the SIF was soon found to be unrealistic given the available resources, and the main outputs sought from the first phase of the trial became narrowed to (i) a process for identifying assets of importance, and (ii) a listing of important assets from a state-level perspective. Cleland (2008 p. 234) noted that the Avon Catchment Council 'was not content with this being the 'end of the road' for SIF', and wanted to move beyond lists of important assets in order to prioritise investments. She observed that the starting point for pursuing a second phase of the SIF (becoming known as SIF2, as against SIF1 for the first phase) was an 'assert[ion] that 'investigations into tractability are a high priority for this project to achieve its goal …'. The SIF Steering Committee's experience with the state-level trial had revealed to its members that 'feasibility information is an important ingredient in determining investment priorities' (ibid. p. 237) (The term 'feasibility' was used as a synonym for 'tractability' and came to be used in its place.) This starting point led to consideration of 'the possibility of expanding the two dimensional value-threat matrix into a three dimensional matrix incorporating information on tractability' (ibid. p. 234).

Due to concerns about the resource demands of obtaining detailed feasibility information for all assets, the final report for SIF1 (Department of Environment 2003) signalled that SIF2 'would employ a filter that focused assessment for feasibility on those assets considered important through the value-threat matrix' (Cleland 2008 p. 237). High-value/high-threat assets were referred to as tier-1 assets, which would receive the greatest level of further investigation. Indeed, it was expected that limited funding would make it unlikely that assets outside tier 1 would be further investigated. Cleland (ibid. p. 237) noted that '[t]he logic for focusing further efforts on tier 1 assets was not reported in any detail by the SIF Steering Committee', but observed that the final report for SIF1 detailed the following grounds for confining further investigation of biodiversity assets to tier-1 assets (ibid. pp. 237-238):

- 'Without further risk analysis it was not considered acceptable to focus on other high-value, but less-threatened assets, as this would effectively assign tier 1 assets to a low probability of retaining the full range of their current biodiversity values.

- Working in some of the more highly threatened areas would be more likely to deliver a better understanding of managing salinity including the development of new technologies.

- Further investigation of tier 1 assets would see the priority of some landscapes downgraded and others elevated'.

Commencement of SIF2 saw a return of David Pannell to attending meetings of the SIF Steering Committee and contributing significantly to out-of-session tasks and dialogue. Work on progressing SIF2 began with defining feasibility. Dr Pannell developed a two-page template on how to assess feasibility, which focused explicitly on the benefits and costs of public investment. Benefits were to be assessed 'in terms of the contribution the option would make to the achievement of the management goals (in terms of recovery, containment or adaptation) and costs were to be in relation to the cost of the option to government or the regional group (ibid. p. 239). The final report on SIF2 was released in December 2006.

A consultancy commissioned by the Department of Catchment and Land Management found around this time that requirements for public funding would still be very large ($950 million over 30 years) even if investment were limited to the highest priority vegetation assets. Pannell (pers. comm., March 2009) recalled that 'this was an extremely important finding, and had a big impact on the [SIF Steering] Committee. … It highlighted that to really achieve outcomes required much more intensive and expensive interventions than programs usually made. And that seed funding and relying on social capital to do the rest was never going to actually protect the highest priority assets'.

5.1.8 SIF's third phase

Cleland (2008 p. 273) noted how:

> … the SIF dropped off the Avon Catchment Council agenda during the latter half of 2003 and early into 2004 with all available human resources channelled into meeting the national directive for the preparation of regional NRM strategies … [T]he value of the SIF was rediscovered at the time when the Avon Catchment Council had moved beyond goal and target setting and was struggling with how to manage trade-offs and set priorities for regional delivery. Despite this, the SIF only partially informed the final Avon NRM strategy and subsequent investment plan.

She reported further that:

> Feedback from the Avon Catchment Council and other NRM bodies has indicated that an 'easier-to-apply' prioritisation framework is required. From this experience, SIF3 [the third phase of the SIF] was developed and piloted in partnership with the South Coast NRM group in southern Western Australia and the North Central Catchment Management Authority in southern eastern Victoria' (ibid. p. 285).

The SIF3 project was co-led by David Pannell and Anna Roberts (nee Ridley). SIF3 advanced SIF1 and SIF2 on a number of fronts, while remaining faithful to the original principles. Pannell (2007) explained that '[b]asically, SIF1 and SIF2 focus almost solely on key assets, and provide a process for evaluating which assets should be the highest priority based on a set of agreed principles. SIF3 broadens the focus to provide guidance about investment in technology development, helps the user select which policy tool to use for a given asset, and embeds a large body of existing research and analysis in the form of decision rules, to make the job much easier for users'. Table 5.3 identifies the key similarities and differences between the different phases of SIF development.

Table 5.3: Key similarities and differences between SIF1, SIF2 and SIF3

	SIF1	SIF2	SIF3
Decision framework			
Asset-based approach	✓	✓	✓
Used key criteria to identify priority assets	Partial	✓	✓
Handles priority of industry development/technology development		Partial	✓
Includes detailed public benefit/private benefit framework, accounting for likely landholder behaviour			✓
Shows how to combine key criteria into an overall recommendation			✓
Recommends case-specific policy mechanisms (e.g., extension vs. incentives vs. research and development)			✓
Uses GIS to consider multiple assets			✓
Embeds a large body of research and analysis to inform the decision rules			✓
Application			
Piloted at state level	✓	✓	
Piloted at regional level	✓		✓
Dedicated project team			✓

Source: Pannell (2007)

A key feature of SIF3 is the distinction it makes between 'localised' and 'dispersed' assets. Localised assets are defined as 'discrete, high-value assets in particular locations', while dispersed assets are defined as 'groups of assets that are spread across the region, such as agricultural land, or the many small parcels of remnant vegetation on farms' (Pannell et al. 2007). Based on a range of economic analyses, it was assumed that '[t]he payoff from successfully investing in well chosen localised assets is likely to be high', whereas the payoff per hectare from dispersed assets was assumed generally to be much lower (ibid.). The distinction was made because high payoffs from investing in localised assets may justify relatively expensive measures like engineering works or high levels of incentive payments to manage those assets, while the low payoffs per hectare from investing in dispersed assets would justify measures involving only a low cost per hectare. It follows that investment in dispersed assets needs to be relatively low-cost per hectare, and highly effective over large areas, to be competitive with investment in well-chosen localised assets (ibid.).

The SIF3 project leaders concluded that the pilots of the framework strongly influenced both the regional NRM bodies involved in them, and that they have 'demonstrated that it is feasible to apply SIF3 to select NRM investments that are likely to yield more valuable environmental and natural resource outcomes than currently. SIF3's focus on outcomes is much needed. Existing programs such as NAP and NHT have encouraged activity and expenditure of funds' (Ridley et al. 2008 p. 3).

Nevertheless, the particular asset-based approach adopted for SIF3 attracted some criticism for its lack of focus on social assets. David Pannell's argument against 'investing NRM funds in social structures and networks just for the sake of having these things' was noted in section 5.1.5. Nicholson (2006 p. 2, as quoted in Cleland (2008)) commented that SIF3 'fails to incorporate the need to nurture participation, to share what is learnt and to empower people to act because they want to. The implementation asset, created through Landcare by investing in all who want to participate, is under

threat because of this approach'. The SIF3 project team responded to such criticisms 'by saying that SIF3 is relevant to specific natural assets, and that decision makers should make a separate decision about how much of their budgets they wish to allocated to targeted investment in assets, and how much in untargeted 'capacity building' etc.' (David Pannell, pers. comm., March 2009).

Nevertheless, Cleland (2008 p. 293) remarked that 'the government and community's concentration on protecting the diffuse 'implementation asset' … is likely the biggest barrier standing in the way of achieving the policy changes to which the SIF team aspire'. Ridley et al. (2008 p. 4) acknowledged that

> … achieving the policy changes to which we aspire remains elusive. Australian governments appear unwilling to mandate the use of SIF3 (or any other framework) as a condition of funding to regional NRM bodies, preferring to support a process of voluntary uptake. We are concerned that regional bodies have few incentives to pursue NRM outcomes cost effectively, but remain hopeful that stronger incentives to do so will be introduced eventually.

5.1.9 The public: private benefits framework

A further important contribution of SIF3 has been its development of a public: private benefits (PPB) framework. This framework distinguishes between four classes of policy tools identified in Table 5.4. It 'provides a simple graphical approach that spells out the logic for selecting the most appropriate class of policy tool for influencing the behaviour of private individuals in cases where their actions have positive or negative impacts on others in the community' (Pannell 2008b). Further, it claims to provide 'a pathway to more cost-effective and scientifically defensible investments in management of dryland salinity by providing guidance on the broad categories of policy measures that are appropriate in different circumstances' (Ridley et al. 2007 p. 15).

Table 5.4: Alternative policy tools for seeking management changes on private lands

Class of policy tool	Policy tools included in class
Positive incentives	Financial or regulatory instruments to encourage change
Negative incentives	Financial or regulatory instruments to inhibit change
Extension	Technology transfer, education, communication, demonstrations, support for community networks
Technology development	Development of improved land management options, such as through strategic R&D, participatory R&D with landholders, provision of infrastructure to support new management options
No action	Informed inaction

Source: Pannell (2008a)

The PPB framework makes it clear that:

> … the choice of policy response depends at least as much on the level of private net benefits from the land-use change as on the public net benefits … . This is an important finding, particularly as some environmental managers focus predominantly on the public benefits, but pay little attention to the estimation of private net benefits. As a consequence, they are under-informed about the

landholders' likely responses to any proposed change in land use, which is one of the key factors that should influence the choice of policy response (Pannell 2008a pp. 238-239).

Nevertheless, application of the framework requires estimation of both private net benefits (or costs) and public net benefits (or costs). Pannell (2008a p. 239) observed that the framework requires little extra effort from environmental managers to estimate public net benefits, since '[t]hey are already choosing which environmental projects are of highest priority, so there must be some assessment of environmental benefits, even if only implicitly. Projects may not necessarily be ranked according to their environmental benefits with great precision, but even relatively qualitative ratings could be applied within this framework. If further precision is required, a range of market or non-market valuation methods may be appropriate …'. He noted that the options available for gauging private net benefits include economic modeling, surveys of farmers or farm advisers, and using a conservation auction as a means of landholders revealing their willingness to act given a subsidy level chosen by themselves.

The recommendations deduced from the PPB framework are:

- Positive incentives should be used where public net benefits[37] of a proposed practice are highly positive and private net benefits[38] are close to zero.

- Negative incentives should be used where public net benefits are highly positive and private net benefits are slightly positive.

- Extension should be used where public net benefits are highly positive and private net benefits are slightly positive.

- No action should occur where private net benefits outweigh public net costs, where public and private net benefits are both negative, where private net benefits are sufficiently positive to prompt rapid adoption of environmentally beneficial activities, or where private net costs outweigh public net benefits (provided that technology development is not sufficiently attractive) (Pannell 2008a).

Analysis with the PPB framework implies 'there should be a number of shifts in emphasis in the funding directions of the existing policy program [for managing dryland salinity in Australia], most notably less emphasis on incentives and extension, and more on plant-based R&D' (Ridley et al. 2007 p. 15).

[37] Public net benefits 'means benefits minus costs accruing to everyone other than the private land manager. They exclude any costs borne by the environmental manager in the process of intervening to encourage the change in land management' (Pannell 2008a p. 225).

[38] Private net benefits 'refer to benefits minus costs accruing to the private land manager as a result of the proposed changes in land management. They exclude transfers which are part of the policy intervention, so that we can compare landholder behaviour with, and without, the intervention. In principle, private net benefits are broader than financial benefits, and include the broad range of factors that influence the relative advantage of the new land use options (as perceived by the landholders) such as riskiness, complexity, social considerations, personal attitude to the environment, and farming-system impacts of the land-use practice …' (Pannell 2008a p. 225)

5.1.10 INFFER beginnings

Pannell et al. (2009a) reported that the regional bodies with which SIF3 had been piloted requested that the project leaders develop a more general framework applicable to environmental threats beyond dryland salinity. Moreover, they wanted this new framework to be developed rapidly enough for it to be used in revising their investment plans. With the existing major national NRM programs (NAP and NHT2) due to expire in June 2008, the project leaders also saw an opportunity to develop a tool that could be used in any new program. Hence, they decided in late 2007 to start immediately on developing an initial version of INFFER. As this initial version became applied by various regional NRM bodies, and the project team (still co-led by David Pannell and Anna Roberts) learnt from their own observations and regional feedback, the framework became more flexible and general.

INFFER is presented as 'squarely focused on achieving NRM outcomes cost-effectively' and as a means of 'compar[ing] cost effectiveness across different asset types and different project sizes' (Pannell et al. 2009a p. 2). These characterisations are consistent with the SIF frameworks, given that '[t]he critical goal of the SIF is cost-effectiveness: the achievement of salinity recovery and containment at least cost' (Black et al. 2004 p. 169). Like the SIF, therefore, INFFER has been presented as a mainstream economic approach to priority setting.

A particular motivation in developing the new framework was, within the confines of maintaining adequate rigour, to keep its application as simple as possible. Pannell et al. (ibid. p. 3) remarked that 'ease of use was particularly important …, as reinforced by the observation that, internationally, there is low usage of many decision tools intended to support decisions about environmental investment'.

5.1.11 INFFER remains asset-based

Like SIF3 and its predecessors, INFFER is an asset-based framework which the project describes as follows:

> … [P]rojects are built around assets. Each project assessed is built around a particular asset or set of assets. The output from the INFFER process for an asset is an assessment of a particular project related to the asset(s), rather than an assessment of the assets per se (Anon. 2009).

A justification for continuing with an asset-based approach for INFFER was:

> Even in large government programs for the environment and natural resource management, the available funding is small relative to the problems it is intended to address. Spreading the available public resources thinly across many areas will result in little or no effective protection of any of the threatened assets. Experience shows that basing decisions around key assets helps to improve the cost effectiveness of investment (Anon. 2009).

INFFER is intended only for projects with a clear focus on protecting or enhancing specific natural resource assets. It is not meant for projects focused on general education, awareness raising, capacity building, or research that is not targeted at specific assets.

Significantly, given the opposition encountered during the time of SIF1 and SIF2 to developing a framework designed to prioritise investments across asset classes, INFFER 'allows systematic

comparison of investment options across all types of NRM issues, not just within any one category (such as biodiversity)' (ibid.)[39].

5.1.12 The process of applying INFFER

Development of a process for applying INFFER recognised 'the constraints of time and resources available for analysis of priorities, such that it was not realistic or efficient to conduct detailed assessments for every possible investment opportunity' (ibid. p. 3). Hence, the process begins with simplified assessment of a long list of environmental assets, and grows more detailed as the list becomes narrowed to the most attractive options. The steps in the process include (Park et al. 2011):

a. Develop a list of significant natural assets in the relevant region(s). At the regional level, the list may include 100 to 300 significant assets.

b. Apply an initial filter to the asset list, using a simplified set of criteria. The list developed in step (a) is filtered down to around 20-40 assets. The suggested approach is to identify assets of high significance, with high current or predicted future damage. This filtered list is then further reduced (to around 10-20 assets) using the five questions on the INFFER Pre-assessment Checklist. Assets may be culled at this stage because they are not spatially explicit, because a specific, measurable time-bound goal cannot be formulated, or because an initial assessment indicates that the project would not be cost-effective.

c. Define projects and conduct detailed assessments of them. The INFFER Project Assessment Form (PAF) is used in this step to develop an internally consistent project for each asset on the reduced list. Using the information recorded in the form, the Public: Private Benefits Framework is applied to help select policy mechanisms, and calculate a Benefit: Cost Index (BCI) to be used in project ranking. A Project Assessment Report is prepared which includes: the BCI and other key information recorded in the PAF.

d. Select priority projects. A short list of priority assets/projects is selected on the basis of information in the Project Assessment Report and other relevant considerations.

e. Develop investment plans or funding proposals (depending on whether INFFER is being used to allocate an internal budget or to develop and assess projects for external funding).

f. Implement those projects that receive funding. It is proposed that the first stage of a project should consist of a detailed feasibility investment, involving targeted collection of additional information to strengthen the assessment performed in step (c).

g. Monitor, evaluate and adaptively manage projects.

Completing the Project Assessment Form (step (c)) for an asset requires information on its value, the main threats to that value, and the degree to which the value of the asset is currently, or expected to be, damaged by those main threats. It also requires information on a 'SMART' (specific, measurable, achievable, relevant and time-bound) goal for the asset and the feasibility of pursuing this goal. Feasibility assessment involves information on: a project proposed to address the threats; likely

[39] Pannell (pers. comm.., March 2009, original emphasis) commented, 'This opposition hasn't totally gone away, but it is MUCH reduced. It just needed someone to demonstrate a practical and logical way to do it'.

reductions in damage from implementing the project (technical feasibility); likely time lag until the project would deliver a substantial share of it benefits; magnitude of potential negative side-effects from the proposed project; adoptability of the proposed project and associated behavioural changes by relevant land or water managers (given the specific policy mechanism/s envisaged for motivating this behavioural change); administrative and political feasibility of implementing the project; costs of the project; and costs beyond the project of maintaining its outcomes.

5.1.13 Calculating a 'Benefit: Cost Index' for a proposed project

The BCI for a project is calculated under INFFER as follows:

$$BCI = \frac{V \times W \times F \times A \times B \times P \times G \times DF_B(L) \times 20}{C + PV(M)} \qquad (1)$$

where

- V = value of the asset in good condition (score out of 100, or non-market valuation);
- W = multiplier for proportionate impact of works on asset value (from 0 to 1);
- F = probability that the project will not fail due to problems with technical feasibility (from 0 to 1);
- A = multiplier for adoption of changed management by private landholders (proportion of the adoption level needed to achieve project goal) (from 0 to 1);
- B = probability that the project will not fail due to private landholders adopting practices adverse to achieving the project goal (from 0 to 1);
- P = multiplier for socio-political risk (probability that socio-political factors will not derail the project, and that required changes will occur in other organisations) (from 0 to 1);
- G = probability that essential funding subsequent to this project will be forthcoming (from 0 to 1);
- DF_B = discount factor function for benefits (proportion between 0 and 1)), which depends on L
- L = time lag until the majority of anticipated benefits from the project occur (years)
- C = short-term cost of project ($ million in total, over 3-5 year life of project);
- PV = present value function to convert future costs to equivalent present-day values; and
- M = annual cost of maintaining outcomes from the project after its completion ($ million per year beyond the immediate project).

The numerator of equation (1) measures the net benefits from a project as discounted for how slowly they are expected to arise. Impact of works (W) specifies the proportionate expected increase in asset value if the project is fully implemented and all required behavioural changes eventuate. Hence, the product V x W gives what the total net benefits of the project would be if it were fully implemented and if all its net benefits occurred immediately.

This level of net benefits is then adjusted downwards according to the socio-economic feasibility of the project measured as A x B x P x G. This adjusted level of net benefits is then further modified downwards according to (a) how long it takes until a large proportion of the benefits is expected to

arise (*L*), and (b) the discount factor chosen to reflect 'time preference', or how '[b]enefits that occur further into the future are a lower priority than similar benefits that occur rapidly' (Anon. 2008c p. 5).

The denominator of equation (1) measures the expected costs of the project and maintaining its benefits. Up-front costs (*C*) are those incurred within the immediate time frame of the project, which is assumed to be short enough that discounting for time preference can reasonably be ignored. Maintenance costs (*M*) are those incurred beyond the life of the project but which are needed to maintain the benefits of the project into the longer term. For instance, such costs may need to be incurred in monitoring and evaluation, enforcement, or in ongoing compensation payments to private landholders. The present value function (PV) discounts this ongoing stream of costs to account for time preference, and aggregates the stream of discounted costs into a present value.

Each of the parameters comprising equation (1) is quantified using data from the Project Assessment Form. When working with regional NRM bodies in converting this assessment to a score for inclusion in the BCI, the recommended standard approach '(for reasons of simplicity and user acceptance) is to score the value of the asset [*V*] relative to an arbitrarily selected reference environmental asset (an asset of high national significance). The reference asset has a V score of 100, and other assets are scored relative to that, with guidance provided in the Instruction Manual. This score incorporates both market and non-market aspects of the asset' (Pannell et al. 2009a p. 7). The score assumes that the 'SMART' goal for the asset is achieved. Table 2 in the instruction manual for the Project Assessment Form (Pannell et al. 2009b) presents a 'Guide to scoring *V* for different types of assets'. This recommends that assets of international significance (e.g., Great Barrier Reef) be assigned *V* scores exceeding 100, assets of national significance (e.g., the Macquarie Marshes) be assigned scores between 50 and 100, and assets of only local significance (e.g., a locally valued wetland or creek) be assigned scores between 0.1 and 2. The instruction manual explains:

> If we were conducting a full benefit-cost analysis, we would attempt to convert the environmental and social values of these assets into dollar terms, using techniques such as choice modelling or contingent valuation. The INFFER scoring scheme is proposed as a simple alternative in the expectation that sufficient information on dollar values will not be available (ibid. p. 60) .

Pannell et al. (2009a) noted that use of this simplified method for scoring asset value means that a resulting BCI value has no obvious meaning in itself, but contended as follows that BCI values calculated for different projects permit them to be ranked: '[The BCI] is able to compare the cost effectiveness of projects across different types of environmental assets and projects of different spatial and temporal scales'(ibid. p. 2). The ability to rank projects focused on different types of natural assets is rare among the decision support tools for project prioritisation that have been developed to date, with most such tools focused on a particular type of asset (e.g., biodiversity in the case of the project Prioritisation Protocol developed by Joseph et al. (2008)).

Given that the BCI equals the benefits of a project (albeit usually measured in non-monetary terms) divided by its costs, it 'is closely related to a Benefit Cost Ratio' (ibid. p. 7). When *V* is measured in monetary terms, indeed, the BCI can be interpreted as a conventional benefit-cost ratio (BCR), with the project deemed economically efficient (in the sense of a potential Pareto improvement – see section 1.4) if its BCR exceeds unity.

The proposed procedure for allocating a fixed budget between competing projects on the basis of their BCIs involves ranking the projects in descending BCI order and continuing to fund projects down the list while the cumulative cost remains within the total available budget (Anon. 2008c). However, '[t]he

BCI is not to be applied in a simple prescriptive way. It provides information that can assist decision makers, but given the inevitable limitations of data, and the likely relevance of other considerations that have not been capture in the assessment, the resulting ranking of options should be treated as a guide, rather than as 'the answer'' (ibid. p. 7).

5.1.14 Progress with, and obstacles to, adoption of INFFER

Pannell et al. (2009a) reported that 19 of Australia's 56 regional NRM bodies had used or trialled INFFER as of January 2010, with most of these trialling its use with a limited number of natural resource assets. They observed that at the state government level interest had been strongest in the two states where the INFFER team had the strongest engagement and environmental policy direction was already towards an asset-based approach. They reported also that INFFER was the only project planning tool recommended to applicants in respect of the 2008-09 round of funding under the Caring for our Country program (Australia's main national NRM program).

Nevertheless, the project team reports that challenges remain ahead in convincing potential users of INFFER that the additional time and resources it requires to apply, compared with the investment priority-setting procedures used previously, is more than compensated by the gains in investment outcomes. They remarked that the change to a procedure based on economic reasoning 'requires support for a change of mindset, and in some cases even a change in organizational culture' (ibid. p. 10). Apparently, some of the regional bodies engaged with the INFFER project 'have been unable or unwilling to devote the time required to complete Project Assessment Forms to a good standard. … We consider five days [completing a form for each asset] to be a reasonable level of due diligence to be undertaken by bodies that are applying for large amounts of public funding, but some individuals who are used to receiving funds with less work are not yet convinced' (ibid. p. 10).

Aside from the problem of entrenched mindsets, cognitive limitations have also been identified as an obstacle to users following the INFFER process with due diligence. Hence, provision of training and support to users is emphasised by the project team, with this provision based on the principle of:

> … empower[ing] users to complete analyses themselves. We have found this to be an extremely important aspect of the project, especially given the low existing capacities of many regional NRM bodies in relation to rigorous planning and prioritisation processes … [O]ur experience has been that no matter how good our written support materials are, many users require interactive support' (ibid. p. 8).

The project team sees the focus of the INFFER process on internal consistency as a strength of the process: '[T]he on-ground actions specified would achieve the goal; the policy mechanisms and delivery mechanisms specified would result in the required on-ground actions; and the costs specified are consistent with the policy and delivery mechanisms' (ibid. p. 4). This focus on internal consistency makes INFFER a valuable project development tool. David Pannell (pers. comm., March 2009) remarked how:

> Some users come to us with a pre-defined project that they want to assess using INFFER. What we inevitably find is that the project makes no sense. They have to adjust the goal, or the works, or the interventions, or the budget, or several of these things, in order for it to make sense. This internal consistency aspect is crucial because without it you cannot validly compare projects. Without it you end up funding the project that has been exaggerated the most.

Regional NRM bodies using INFFER are reported to 'have often been led toward substantial changes in their investment priorities', and the project team noted that this may lead to political obstacles given that 'INFFER may recommend investments that conflict with existing political preferences' (Pannell et al. 2009a p. 10).

A further obstacle to acceptance of INFFER has been that 'some users wish to define 'the community' as one of their NRM assets' (ibid. p. 10). The project team argued in response that:

> The community is central to the INFFER process, but it is not appropriate to treat it as an asset in the same way as we define a wetland or river as an asset. We assume that the purpose of the public funding is to improve environmental and natural resource outcomes, and while the community plays a number of essential roles in that …, we are not investing in the community for its own sake. (There are other government programs that do that.) Rather the program would support the community to pursue environmental and natural resource outcomes that are important to the community (Anon. 2009).

5.2 INFFER: a perspective from 'outside'

The literature on INFFER, and also the SIF, has been authored almost exclusively by members of the project teams associated with these frameworks. The comments below offer a perspective on this framework from the 'outside'. This perspective is informed by the reviews of BCA, MCA and deliberative methods of evaluation presented in chapters 2, 3 and 4, respectively.

5.2.1 Issues with a scoring-based approach to valuing assets

Given that the method of calculating a BCI is based on benefit-cost analysis, the problems with BCA discussed in Chapter 2 apply to the BCI to the extent that the actual procedures used in calculating it are equivalent to those used in conventional BCA. When *V* is measured monetarily by applying a non-market valuation (NMV) method, for instance, the concerns over monetising environmental values discussed in section 2.2.2 become relevant, as do the specific criticisms of NMV methods discussed in section 2.2.4.

Nevertheless, the standard BCI method potentially offers ways around some other criticisms of BCA. It is more practical and affordable than the conventional BCA method (see section 2.2.8), and thus more likely to be applied effectively at community-based and other decentralised levels of governance. The INFFER process within which BCIs are calculated was designed to encompass a process of community consultation, and thus it provides scope to address criticisms of BCA due to the top-down manner in which it is normally applied. This consultative process allows stakeholders 'to express values and preferences for different NRM assets [and] to provide local knowledge about assets and their management …' (ibid. p. 9). Hence, it offers potential to address criticisms of BCA for valuing environmental and natural resource assets using the principle of individual sovereignty rather than on the basis of deliberation among affected parties[40] (see section 2.2.5).

Scoring-based measurement of the parameters used in valuing project benefits allows factors to be accounted for that often are excluded from BCA studies due to problems in assigning monetary values to them. The implications of transaction costs for the likely benefits accruing from a project are not

[40] David Pannell (pers.comm., March 2009) responded to this statement as follows: 'Strongly agree. The values of our assets are very much a reflection of some sort of community process'.

normally accounted for in BCA studies (see section 2.2.7), for instance, but the BCI procedure of assigning scores to *A* and *P* (associated with socio-economic feasibility) offers a practical means for these implications to be incorporated.

Scoring-based methods of valuation are prone to the kinds of criticisms often levelled by mainstream economists at MCA methods. One such criticism is that scoring lacks the so-called objectivity of valuing project effects using market prices on NMV methods, thus opening up the valuation process to subjectivity which mainstream economists tend to expect will be abused for rent-seeking purposes. David Pannell (pers. comm., March 2009) commented on this issue as follows: 'Yes, the process of selecting a value score for the asset is subjective. However, it just makes explicit what we already do implicitly, so it can be scrutinised and the decision process can be transparent'. He remarked further that, 'We consider a system of quality assurance/expert review of assumptions to be essential (not just for INFFER but for any prioritization process where people put in proposals'.

Even without risks of abuse by rent seekers, however, scoring-based evaluation methods face challenges in convincing relevant clientele of the credibility of the scores that have been assigned. The key challenge of this kind in applying the BCI for the purpose of ranking different projects lies in scoring *V* for each of their targeted assets such that the scores are accepted as credibly reflecting the relative values to the community of those various assets.

The standard procedure for calculating a BCI uses a reference asset to 'calibrate' the process of assigning *V* scores to the various targeted assets. The value of the reference asset is set at 100, and *V* for the asset in question is scored depending on the extent to which its value is assessed to be higher or lower than the reference asset. This is an innovative response to the challenge of strengthening trust in a scoring-based approach to valuation. However, the degree to which this response actually results in *V* scores being assigned more credibly needs to be evaluated. It is possible, for instance, that the choice of reference asset may bias the scoring process in favour of assets similar to the reference asset.

Another likely criticism from mainstream economists is that a scoring-based approach to valuation provides scope for analysts and decision makers to move outside the method of neoclassical welfare economics which most such economists regard as imparting essential rigour to the valuation process. Notably, for instance, the standard BCI procedure provides scope for valuation of an asset based on community deliberation rather than, as required by neoclassical-economic principle of individual sovereignty, by aggregating the individual preferences of community members. The INFFER project team is clearly comfortable with this departure, discussing how '[w]e capture community valuation of various assets in community workshops …' (Anon. 2009). Although open to criticism from mainstream economists, this departure should be welcomed by ecological economists and others arguing for greater use of deliberative processes in methods for valuing environmental goods and services. In any case, use of non-market valuation (NMV) as an alternative to INFFER's scoring-based approach is often impractical given that the required NMV studies have typically not already been undertaken and it is normally too expensive to conduct such studies where the range of investment options to be prioritised involves a large number of assets.

5.2.2 Allocating an investment budget between 'scaleable' projects

It is often the case that a certain kind of project (e.g., protecting remnant native vegetation in a particular locality) can be undertaken across a range of scales, and the funds available to invest in it are likely to be insufficient to undertake it at the upper end of this range. Nevertheless, INFFER requires the decision maker/s to specify in advance the particular scales at which it is planned to undertake each

of the competing projects, in order that the benefits and costs of each can be estimated. The priority-setting framework then ranks these particular versions of the projects. The conventional economic procedure is then to proceed down the ranked list, committing to fully fund each project until the point is reached where the available budget is fully allocated. At this point, however, it is not uncommon for the decision maker/s to want to reconsider how they initially specified the scales of the highest-ranked projects. If project C comes to be ranked highest, for instance, the decision-maker/s might sensibly question whether it should be redesigned at a larger scale than initially specified, with a greater demand on the available budget, and whether the scales and budgets of lower-ranked projects should be reduced accordingly.

This issue was discussed in section 2.2.9 in relation to the use of BCA results, and particularly the NPV/I measure, for informing how a fixed investment budget should be reallocated between competing projects. It was concluded there that NPV/I measures calculated for different projects might validly be used as a guide to the *directions* in which an available budget should be reallocated between projects, but that these measures cannot, except in one extremely unlikely scenario, be used validly as a guide to the *degree* by which reallocations should occur. The exception is the case of constant returns to scale, where the NPV measures for all projects increase or decrease at the same rate as their *I* values (i.e., their respective scales). In this highly unlikely case, economic efficiency would justify allocating the entire available budget to the project ranked highest by the NPV/I criterion. Other than in this exceptional case, identification of the optimal budget allocation between competing projects ideally requires every project to be evaluated at each of its feasible levels of scale. Given the costliness of pursuing this ideal, however, short-cut approaches will normally be required. As noted in section 2.2.9, the short cut suggested by Alston et al. (1995) in the context of investing in agricultural research and development was to evaluate three alternatives for each type of project: the project at a 'baseline' scale and at *X* per cent higher and lower than this scale. NPV/I measures for projects scaled between the baseline and alternative levels might then be approximated roughly through interpolation.

The INFFER process seems flexible enough to accommodate a short-cut process of this nature[41]. Such accommodation would better enable INFFER to identify the allocation of available NRM funds between projects that maximises economic efficiency. Nevertheless, the additional costs of accommodating such a short-cut process of allowing for scaleable projects would need to be weighed against the increase in economic efficiency expected to result.

5.2.3 Accounting for the benefits of integrated environmental management

The early 1980s saw Australian governments acknowledging the interdependence of different environmental assets, threats and solutions through integrated programs of management (e.g., Watson et al. 1983). This acknowledgement led to promotion of the concept of integrated catchment (or environmental) management, founded on a systems approach to natural resource management (Marshall et al. 1996).

This recognition of the advantages of managing interdependent environmental issues in an integrated manner remains central to current programs of natural resource management at national and state levels. For instance, the Caring for our Country Business Plan 2009-2010 Business Plan seeks 'integrated projects' (Commonwealth of Australia 2008a, figure 1), particularly in respect of '[l]arge and medium projects [that] can integrate actions across large geographic areas and key assets and

[41] Pannell (pers. comm., March 2009) remarked, 'We encourage this. We are currently demonstrating how it would work in action …'.

achieve significant multiple benefits[42]' (ibid. p. 23). Similarly, a required outcome specified in the NSW Government's Standard for Quality Natural Resource Management is: 'Management of natural resource issues at the optimal spatial, temporal and institutional scale to maximise effective contribution to broader goals, deliver integrated outcomes and prevent or minimise adverse consequences' (Natural Resources Commission 2005 p. 8). One of the steps suggested for achieving this outcome is: 'Evaluate the potential for delivery of multiple benefits – environmental, social and economic' (ibid. p. 8). This required outcome is reflected in many of the Catchment Action Plans (CAPs) developed by Catchment Management Authorities (CMAs) in NSW. In the CAP developed by the Border Rivers – Gwydir CMA, for instance, one of the stated principles is 'maximise the benefit of a project by ensuring that actions taken can provide benefits to more than one target where possible' (Border Rivers - Gwydir Catchment Management Authority 2006).

Despite this ongoing emphasis, regional NRM bodies face continuing difficulties in evaluating interdependent projects in an integrated way or, equivalently, accounting for the multiple benefits of any particular project (particularly when secondary benefits would accrue outside the program primarily responsible for the project). For instance, an officer with the Border Rivers – Gwydir CMA commented when interviewed for this project in 2008 that:

> In the Catchment Action Plan, we said that it [integrated catchment management] happens, that investment in one thing can promote something else. But we haven't developed a mechanism for measuring it, or integrating it. Our investments to date have been quite separate (Marshall 2008b p. 22).

Asked how the philosophy of integrated environmental management was applied by his CMA when evaluating investment options, an officer of the Northern Rivers CMA answered, 'With difficulty, actually' (ibid. p. 63). An officer with the Namoi CMA commented likewise, 'The principle of integrating natural resource management is there, but the capturing and measuring of it, and how you do it, is our big gap' (ibid. p. 47). These responses are consistent with the finding of the (NSW) Natural Resources Commission (2008b p. 5) that:

> … few audited CMAs are consistently prioritising projects that provide multiple NRM benefits, as required by the Standard [for Quality Natural Resource Management]. Rather, they more commonly take a 'siloed' approach, which means they direct their investments to improving specific aspects of the landscape (such as native vegetation or soil) without systematically considering the potential to generate multiple assets across the landscape. As a result, their investments may not be making the greatest possible contribution to the statewide targets.

In their review of priority-setting tools for the SIF, two colleagues of David Pannell at UWA's School of Agricultural and Resource Economics expressed doubts regarding the consistency of an asset-based tool with a commitment to integrated management. They were concerned particularly that an asset-based approach 'may miss complementarities in the on-ground treatment' and thus undermine efforts to account for linkages between different natural resource problems through an integrated approach to management (Black et al. 2002 p. 47).

[42] The NSW Government's Standard for Quality Natural Resource Management defines multiple benefits as 'outcomes that occur when management actions deliver benefits across institutions, spatial areas, resource assets, time scales and interest groups within the community' (Natural Resources Commission 2005 p. 4).

These economists proposed that a treatment-based approach instead be adopted for investment priority setting under the SIF, with 'treatment' defined as 'a system of integrated management that addresses the threat of salinity to one or more assets …' (ibid. p. 47). For instance, a particular incidence of salinity may threaten two valuable assets: a wetland and a road, and a treatment may be available (e.g., revegetation of a sub-catchment) that protects both of these assets.

Black et al. (ibid.) were concerned particularly with the path-dependency implications of starting an investment priority-setting process by identifying high-value assets and then proceeding to define projects addressing threats to these assets. They argued that this choice of starting point is likely to exclude from consideration projects with a more integrated focus that generate a high level of aggregate benefit across a range of assets but only a medium level of benefit for any particular asset.

This is not to deny that an asset-based process is capable of defining integrated projects for each high-value asset, or group of assets, identified at the outset of such a process. For instance, INFFER documentation states that '[a] project might be based on actions that will protect or enhance a group of assets that are located in a cluster, or a set of similar assets that are in different places but require similar management', and also 'an asset could be a single localised thing (for example, a particular wetland or river), or it could be a collection of smaller assets, such as remnant vegetation on farms in a region, or agricultural land in a region' (Anon. 2009). The point, however, is that the integrated projects defined under such a process for a high-value group of assets will not always be those that generate the greatest investment returns when consequences for all assets are taken into account[43].

Accordingly, these economists argued that a treatment-based approach 'is likely to be a more effective way of identifying projects that generate integrated benefits, as opposed to the asset approach' (ibid. p. 48). They noted that 'identification of treatments would require the integration of asset identification and risk assessment across all assets at an early stage. … For the purposes of the prioritisation process, initially a single asset may appear within a number of potential treatments. Thus, potable water supplies within a catchment may be potentially protected by catchment wide distribution of vegetation, or point treatment. … The task then is to prioritise treatments' (ibid. p. 48). They recognised that this approach would likely require inter-agency cooperation at an earlier stage than is required with an asset-based approach.

The INFFER project team's position on integrated environmental management has been explained as follows:

> What we steer away from is projects that try to achieve things at the whole-catchment scale that are actually infeasible at that scale given the available resources. Most of the words written about intervention and multiple benefits at the catchment scale are quite unrealistic, and for this reason are counterproductive. They tend to push the effort towards investments that are less effective, not more, because they cause people to spread funds thinly, or to prioritise extension (in order to

[43] David Pannell (pers comm.., March 2009), responded to this statement as follows: 'That is true, but in my judgement the benefits of a more focused asset-based approach outweigh the costs in practice. Even in theory, the additional benefits of moving from an asset-based to a treatment-based approach are small – much smaller than the benefits from going from the old system to INFFER. And in practice, the additional benefits may be negative if it leads people back to spreading out the money too thinly. The asset-based approach is better from a psychological perspective – it seems to help people focus more clearly on outcomes. Given how poor decision making and project design has been up to now, we are not aiming for perfection. Realistically, I hope we can take Australia from a situation where little of the NRM money was spent in a way that achieves enduring and highly valuable outcomes, to a situation where some funded projects do achieve worthwhile outcomes.'

> reach as many people as possible) without considering the adoptability of the required works. … We actually take a highly integrated focus, but limit the scale to what the resources can actually afford. That is part of our focus on internal consistency (David Pannell, pers. comm., March 2009).

The reluctance of the INFFER project team to adopt a treatment-based approach to investment priority setting stems from a fear that such an approach will lead to the limited funds available being spread so thinly across multiple assets that threats to none of them are addressed effectively. However, a treatment-based approach that is well-designed would not recommend projects that deliver such an unsatisfactory outcome. Benefits from a project in respect of any asset would be counted only to the extent that the project could reasonably be expected to lead to an increase in the asset's value. Such a well-designed process would recognise, for instance, that realising benefits from mitigating an environmental threat often requires a 'critical mass' of investment and that ongoing 'maintenance' investments are often also required in order to sustain such benefits. Such a process would retain the focus of the asset-based approach on outcomes, and would do so by designing each project so that it is capable realistically of achieving the 'SMART' goal set for it.

The choice ultimately between asset- and a treatment- based approaches to investment priority setting in terms of aspirations for integrated environmental management depends on how successfully asset-based approaches do actually help people to focus more clearly on outcomes, and on the degree to which the ideal forms of each need to be compromised in the process of developing methods that can be applied successfully in real-world contexts. These are empirical questions in need of further research.

5.2.4 Accounting for the consequences of present projects for the ongoing social feasibility of environmental programs

As noted in section 5.1.14, 'community' capacities to help solve environmental problems are not defined as assets under INFFER. Echoing David Pannell's argument against including 'social assets' as a category of assets under the SIF (see section 5.1.5), the documentation for INFFER assumes 'we are not investing in the community for its own sake' (Anon. 2009).

INFFER *is* designed to account for community capacity-building (CCB) activities required within a project to achieve its goals, and to account for existing community, and other socio-economic, capacities when assessing the socio-economic feasibility of the project. However, it is *not* designed to account for the consequences of a present project (whether from specified CCB activities or other aspects of a project) on the socio-economic feasibility of subsequent projects (as would be accounted in evaluation of those projects by the values assigned to the parameters A and P in equation (1))[44]. These consequences may be for community capacities (e.g., for landholders' likelihood of adopting practices that may be promoted to them in subsequent projects) or they may be for the capacities of other parties whose cooperation could be required (e.g., for the likelihood of politicians or government agencies adopting the changes needed for success in subsequent projects).

To be concerned with the consequences of present activities for the kinds of social capacities needed subsequently to make further progress against longer-term goals for environmental policy is not to be

[44] David Pannell (pers.comm., March 2009) responded to this statement as follows: 'This is one of a number of simplifications that we make. It's a balancing act. … There are other sorts of second round effects that we haven't accounted for either. E.g., a project might generate knowledge or technologies that are useful in subsequent projects, or might lead to improved institutions'.

concerned with these capacities 'for their own sake'. Rather, it is to be concerned with the value of these social capacities as 'intermediate goods' on which such progress depends[45]. A given stock of social capacities, or 'social capital'[46], can typically be put to work on a whole variety of collective endeavours (Putnam 1993). Trust and reciprocity arising between landholders and a regional NRM body in one project, for instance, is likely to predispose those landholders to cooperate more voluntarily with the regional body's subsequent initiatives than would otherwise be the case.

The value of social capacities as intermediate goods can be understood as a type of what Weisbrod (1964) first identified as 'option value'. Option values comprise a category of what environmental economists now refer to as 'total economic value'. Option values refer to the value of maintaining some good or service for the benefits it may eventually provide, depending on how an uncertain future unfolds. The option value of a good can be understood as the insurance premium one may be willing to pay to ensure the supply of the good later in time (Dziegielewska et al. 2007). Although environmental economists tend to employ this category of value mainly for environmental and other biophysical goods (e.g., in respect of preserving genetic materials to ensure the option of accessing these goods in the future), it is equally applicable to social capacities.

All projects are likely to affect ongoing social capacities to some extent, either negatively or positively. For instance, some projects are designed and/or implemented in such a way that social capacities of prospective partners in future projects are reduced (e.g., by undermining their trust, reciprocity and thus preparedness to cooperate with those projects). Other projects may increase the social capacities of prospective future partners by engaging them effectively, empowering them, and strengthening their motivation to help implement subsequent projects.

Australian governments recognise that making progress in this domain is often a long-term endeavour (Curtis et al. 2008). They also generally acknowledge the crucial importance of making this long-term endeavour feasible by strengthening and sustaining the community and other social capacities needed to make affordable the transaction (including political) costs that often threaten to stifle momentum in this domain. It has been observed that one of the assumptions underpinning investments by Australian governments in community-based approaches to NRM has been:

> Given [Australia's] small tax base, the continental scale of NRM issues, and limited commitment from urban Australia to environmental issues in the agricultural sector, there are not sufficient resources or knowledge for government to directly manage these landscapes (ibid. p.4).

These authors noted further that: 'Governments have invoked a variety of policy instruments, but there has been heavy reliance on the actions of private landholders and other volunteers to achieve NRM outcomes' (ibid. p. 3).

The (NSW) Natural Resources Commission (2008b p. 9) commented that 'NRM is a long-term process and maintaining the community's trust and ownership requires long-term continuity in the state's model and funding for NRM' (ibid. p. 3), and argued also that:

---

[45] Economists define intermediate goods as goods that are used as inputs in the production of other goods (O'Sullivan et al. 2003).

[46] Coleman (1990) characterised social capital as inhering in the structure of inter-personal relations, yielding value by enhancing individuals' abilities to further their interests.

> As 89% of land in NSW is managed by private landholders, NRM policy responses must include a significant focus on supporting voluntary stewardship. … Much of NRM is about helping landholders and other community groups to help themselves … This voluntary participation and on-ground action can be supported by state and national legislation, policies and programs, but the most effective mechanisms for co-opting voluntary action is at the local or regional level.

The Caring for our Country Business Plan 2009-2010 (Commonwealth of Australia 2008a p. 20) states along similar lines that '[t]he Australian Government recognises both the need for many different groups to work together to bring about change, and the importance of the diverse range of stakeholders working and volunteering in natural resource management across Australia'. It continues: 'We are seeking to maintain [the] involvement of [community-based organisations and groups] into the future. … These groups provide an excellent conduit for the delivery of particular types of projects or elements of projects that contribute to Caring for our Country targets …' (ibid. p. 21).

Regional NRM bodies also acknowledge that success in delivering the NRM strategies for their regions will depend on ongoing development of community and other social capacities. For instance, an 'overriding value' that guided development of the CAP for the Border Rivers – Gwydir NRM Region was as follows: 'Without the support of, and engagement with, local communities, there will be no long-term improvement in resource management with a balance between the environmental, economic and social outcomes' (Border Rivers - Gwydir Catchment Management Authority 2006).

Given the recognition by Australian governments and regional NRM bodies of the value of strengthening longer-term social capacities for environmental management, economic evaluation of investments in this domain needs to account for the implications of investments for ongoing social capacities to be consistent with the stated aspirations of these organisations. This would not involve valuing the implications for social capacities across all dimensions of collective endeavour. Rather, it would entail valuing only the subset of these that corresponds with the scope of the relevant environmental program/s.

The following remark from the INFFER project team indicates it did not consciously decide against accounting for the consequences of non-CCB-focused projects on ongoing social capacities: 'To be honest, it is not an issue that we have discussed, and it is not an issue that anybody else has ever raised with us. People who are interested in capacity building are generally focused on projects which have the sole (or main) aim of building capacity' (David Pannell, pers. comm., March 2009).

It might be argued that the consequences of non-CCB-focused projects for ongoing social capacities are minor for most projects, so that extending the framework to account for them would not justify adding further complexity to the evaluation process. At least two responses can be made to this argument. The first follows from the observation that social capacities might be slow to develop but they are often destroyed quickly (Ostrom et al. 1994). A single project that rides roughshod over the social mores of those it affects, or those who are expected to help implement it, can do serious and long-lasting damage to the prospects of gaining ongoing cooperation from these prospective partners and from those they communicate with.

A second response is that although the contributions of individual non-CCB-focused projects to building ongoing social capacities may each be only minor to moderate, the accumulation of these individual contributions be important for the ongoing social feasibility of environmental programs. Systematic attention to the likely 'social-capacities legacy' of proposed non-CCB-focused projects

when evaluating them may therefore brighten markedly the prospects of eventually accumulating the social capacities required for long-term success in environmental management.

The need for the social-capacities legacy of individual projects to be accounted for *systematically* in economic evaluation of individual NRM investment decisions is highlighted by the economist Alfred Kahn's (1966) analysis of 'the tyranny of small decisions'. Kahn observed that the optimal solution for many social problems can only be identified at a level matching the scale of the problem, but that in many cases no solution is consciously decided at this level. Instead, a series of small decisions is made at lower levels, and the outcomes of these small decisions not uncommonly accumulate to an overall situation that no one intended and few would have wanted. The relevance of Kahn's analysis for environmental management was highlighted by Odum (1982 p. 728), who lamented, for instance, how '[t]he ecological integrity of the Florida Everglades has suffered, not from a single adverse decision, but from a multitude of small pin pricks'.

The point to be made here is that the investment in social capacities needed for longer-term success in environmental management is unlikely to eventuate unless the implications of NRM investment decisions for ongoing social capacities are accounted for systematically when evaluating these decisions. It is possible using the current version of INFFER to account qualitatively for positive and negative implications of projects for ongoing community, or other social, capacities. Given that equation (1) lacks parameters designed to measure the positive or negative side-effects of a project, however, it is not possible under the current version to account quantitatively for how a project may affect ongoing social capacities. Marshall (2010b; 2010a, 2011) developed and trialled a supplement to INFFER's PAF that was designed to obtain the information required to account quantitatively for such 'capacity spillover' effects of projects when prioritising them. However, use of this PAF supplement was found by those trialling it to add considerably to the complexity of applying the INFFER approach.

5.2.5 INFFER and multi-criteria analysis

Pannell et al. (2009a p. 7) reasoned that the INFFER process differs from an MCA approach to prioritising investments in NRM, giving as an example of such an approach the assets, threats and solvability (ATS) model described by Hajkowicz et al. (2006a) and discussed in section 3.5.1 of the present document[47]. They argued that the distinction arises because the BCI calculated in the INFFER process 'is structured in a particular way to address cost effectiveness', while the criteria used in an MCA approach 'are usually combined linearly, and so cannot accurately quantify cost effectiveness' (ibid. p. 7). It is true that Hajkowicz et al. (ibid.) favoured a linear-additive algorithm for operationalising their ATS model (see section 3.5.1), although, like INFFER, the criteria scores for this model could feasibly be combined using a multiplicative algorithm. More generally, the MCA approach is just as open to use of multiplicative algorithms for combining criteria scores as it is to use

[47] Some shared history may explain why INFFER (following on from its predecessor the SIF) and the ATS model both structure environmental decisions in terms of assets, threats and feasibility (or, equivalently, solvability). Cleland (2008) noted that a report by Hajkowicz (2002), the lead author of the article on the ATS model, on the use of MCA for regional priority-setting in Queensland was discussed at the December 2002 meeting of the SIF Steering Committee. This meeting resolved to explore the use of an MCA model within the SIF. Cleland (2008 p. 212) reported also that Hajkowicz made presentations in March 2003 to the SIF Steering Committee and SIF technical working groups to provide 'guidance on developing a standard set of criteria for MCA'. It was noted in section 5.1.6 of the present document that David Pannell, co-leader of the INFFER project team, supported the use of MCA at the time of SIF1 on the basis that it is 'systematic, transparent and participatory'. Moreover, Hajkowicz et al. (2006a p. 88) reported that the ATS had been used 'in Western Australia (Government of Western Australia 2003) to target funds under the Natural Heritage Trust and the National Action Plan …'.

of linear algorithms. Multiplicative algorithms are deemed appropriate for MCA models when the criteria to be combined are non-compensatory (see section 3.4.1).

One of the actual reasons an INFFER process focused solely on the BCI differs from an MCA approach is that the parameters from which the BCI is calculated are not criteria as would normally be defined in an MCA model; i.e., criteria assigned weights that reflect their relative values to decision makers (see section 3.2.1). The parameters specified in equation (1) are defined in such a way (as proportions or probabilities) that weighting becomes redundant. The BCI for a project is calculated simply by inserting values for all the parameters into equation (1).

Nevertheless, the INFFER project team has emphasised that they do not intend decision makers to use the BCI for a project as the only criterion by which it is ranked. They comment that the BCI 'provides information that can assist decision makers, but given the inevitable limitations of data, and the likely relevance of other considerations that have not been captured in the assessment, the resulting ranking of projects should be treated as a guide, rather than as 'the answer'' (Anon. 2008c). However, no guidance is given to decision makers on how they should weigh up these other criteria against each other and the BCI criterion when settling on a final ranking of projects.

An MCA approach could add structure and transparency to this process. BCI values would constitute one of the criteria under this approach, with separate criteria defined for each of the other relevant (e.g., equity, sustainability, employment, legitimacy) considerations. Each of the criteria would then be weighted to reflect their relative values to decision makers. The BCI method of the INFFER process would then become integrated within an overarching MCA model. This suggestion is similar to that made by the economists Alston et al. (1995 p. 484) in respect of evaluating agricultural research programs; namely, that multi-criteria scoring models be developed in which '[t]ypically three or four indices of research contribution may be used: one for efficiency and several for distributional objectives'. The strategy of integrating conventional economic efficiency (i.e., BCA) measures into an MCA model has also been proposed by other economists (Strijker et al. 2000; Sugden 2005), as well as by non-economists (Messner et al. 2006; Munda 1995; Nijkamp et al. 1977; van Pelt et al. 1990) (see section 3.4.2.1).

The INFFER project team has distinguished its BCI method from that of MCA also on the basis that the former:

> … specifies in advance which information is relevant to the decision, provides detailed guidance about the collection of that information, and integrates the information in a specific way that makes the result much closer to BCA than is usually achieved in an MCA. It is quite possible to choose to omit crucial information in an MCA, or to include it in an inappropriate way, but the more structured and guided approach provided by INFFER does not allow this (Anon. 2009).

This is correct given that the MCA approach normally allows decision makers considerable discretion in choosing the criteria against which alternatives will be compared, subject to the MCA analyst striving to ensure that the criteria chosen meet the desirable qualities listed in section 3.2.3 (i.e., complete, operational, decomposable, non-redundant, and minimal). The MCA analyst may also seek to structure logically the process of identifying criteria, in order to reduce the risks of falling short of these qualities. This is the case with the ATS model of applying MCA to environmental decisions, where decision makers are guided to choose criteria under the pre-specified categories of assets, threats and solvability. Even with the ATS model, however, decision makers are left substantial discretion in defining detailed criteria relevant to their particular context.

In contrast, all the 'criteria' (parameters) in equation (1) by which the BCI for any project is calculated are pre-specified by the INFFER team. Although decision makers could add additional parameters (e.g., a parameter to account for any positive side-effects of a project), they are not encouraged to do so. The parameters, and the algorithm by which they are combined, are chosen with the overt purpose of deriving a project performance measure that is as consistent as possible with economic efficiency as defined in neoclassical welfare economics. The parameters are defined deliberately as general concepts (e.g., negative side-effect of actions), leaving the decision maker (with assistance from guidelines provided by the INFFER team) to translate those concepts to each particular decision context. This strategy is similar to that followed by Alston et al. (1995) when formulating a net efficiency index to be used as a short-cut method for evaluating agricultural research programs on the basis of economic efficiency (see section 3.4.2) .

Success with this strategy depends on the capacities of decision makers to undertake with adequate proficiency all the translations required between the general concepts represented by the pre-specified parameters and the specifics of the decision context at issue. In scoring *V* (value of the relevant asset/s) on a scale of one to a hundred, for instance, the decision maker needs to think about the relevant attributes of the asset/s and weigh them up before assigning a score. Where a proposed project will affect multiple assets (e.g., a wetland and a woodland), with multiple attributes of each asset being affected (e.g., ecosystems A and B within the wetland, and ecosystems C and D within the woodland), this becomes a complex task which might be unpacked usefully using an MCA approach in order to be handled in a more structured and transparent manner.

As a further example, assigning a value to *P,* which accounts inter alia for the probability that other organisations will adopt the changes needed for implementation of the project, involves the decision maker identifying and processing a range of information including: (a) identity of the relevant organisations, (b) the changes required from each organisation for project implementation to succeed, (c) the relative importance of each required change for successful project implementation, and (d) the probability that each of these required changes will occur. This amounts to a complex task which the INFFER approach does not deal with in a structured way. To the extent that INFFER lacks a process for making explicit the key assumptions made in assigning values to parameters such as *P*, its contribution to enhancing transparency and accountability in NRM investment decision making is reduced. Deficiencies of this kind could be addressed by elaborating the framework, albeit with the risk of alienating decision makers who perceive it as becoming more onerous to apply. Looks can deceive, however, and judicious elaboration of the framework, perhaps using an MCA approach, could make it easier to apply by enabling the 'unpacking' of complex questions into simpler ones that decision makers can more readily answer.

5.2.6 Structuring and deliberation

It should be noted that efforts to structure a decision process, by proposing the use of INFFER or any other decision framework, invariably impose on the process the values and judgements of those doing the structuring. The more that a framework structures decision making, therefore, the less scope is likely to remain for the framework to accommodate the values and judgements of particular decision makers or those they are representing. For instance, including the parameter *N* in equation (1) to account for the negative side-effects of a project, while not accounting for the project's positive side-effects, reflects a one-shoe-fits-all judgement that the benefits for decision-making quality of including the latter parameter are normally outweighed by the costs – and excludes the option of decision makers deciding this for themselves.

The INFFER process does 'highlight three roles for the community in the [INFFER] process: to express values and preferences for different assets, to provide local knowledge about assets and their management, and to help deliver works and management changes required to protect or enhance the asset' (Pannell et al. 2009a p. 9). However, the examples above indicate how opportunities for community input to the process are circumscribed by the way the process has been structured. Limiting the scope for decision makers to inject their own value judgements into a decision-making process runs the risk of rendering the decisions reached less legitimate in the eyes of themselves and those they represent, and thus of increasing the transaction (including political) costs of getting those decisions implemented effectively.

The point to be made here is that structuring a decision process may confer benefits, but it is likely also to cause costs. The choice in developing INFFER or any other decision framework of how closely to structure the decision-making process therefore involves balancing the marginal benefits of further structuring against the marginal costs. David Pannell (pers. comm., March 2009) observed in respect of this point that:

> … there is a balance to be struck in terms of providing structure and guidance vs allowing flexibility. There is really a lot of flexibility allowed in applying INFFER, but not in crucial aspects. You can't just choose to ignore technical feasibility … A framework that allowed this would be worthless. … [Given the] existing very low capacities of NRM decision makers in relation to integrated assessment, the provision of structure and guidance is quite crucial. It's partly about education, not just about elicitation. … [Moreover] the approach has to be acceptable to Treasury. … Having plenty of structure mandated helps give them comfort that the analysis will be consistent with BCA.

5.3 Discussion

The present version of INFFER represents the culmination of an effort spanning around 11 years to inject an economic way of thinking into how choices are made between different investments that could be made in protecting or restoring Australia's natural environment and resources. This effort has involved major undertakings not only in developing a priority-setting framework but also in engaging with policy and decision makers at different levels (federal and state governments, and regional NRM bodies) to ensure that their respective needs are met sufficiently for them to endorse or apply the framework. The effort has yielded significant success in terms of governments and regional NRM bodies 'buying in' to the framework. This success is all the more notable since '[m]any tools, models and frameworks have been developed to assist with the spatial targeting and prioritisation of environmental investments' (Pannell et al. 2009a p. 1), but 'internationally, there is low usage of many decision tools intended to support decisions about environmental investment' (ibid. p. 3).

The focus in this chapter on INFFER and its predecessor frameworks – SIF1, SIF2 and SIF3 – was motivated by this conspicuous success, and by the consequent likelihood that adoption of any priority-setting process emerging from the present project will depend on its ability to complement or compete with INFFER. Accordingly, this focus stemmed from recognising a need to learn from the experience of developing INFFER, and understand the logic generally behind its development and specifically behind its detailed formulation, before proposing enhancements, modifications or alternatives.

Consideration of INFFER in this chapter benefited from the insights gained from the reviews of the BCA, MCA and deliberative approaches to evaluating environmental funding priorities that were

presented in chapters two, three and four, respectively. While the various phases of SIF referred to the value of injecting an 'economic way of thinking' into the priority-setting process for environmental funding (Black 2008 p. 29), INFFER documentation refers more specifically to introducing a 'BCA mindset' into this process. To the extent that INFFER is applied in practice consistently with the theory of neoclassical welfare economics underpinning BCA, the discussion of the BCA method in chapter two is relevant to INFFER.

Nevertheless, the standard procedure for applying INFFER (involving scoring-based valuation of assets) represents a fairly pragmatic translation of conventional BCA theory to the decentralised context of Australia's regional delivery model for environmental management. Given the emphasis (continued in the Caring for our Country program) of the regional delivery model on collaboration with communities and other stakeholders, the usefulness of economic decision tools in this context depends partly on their capacity for application in a collaborative and consultative manner, compared with the expert-driven mode in which BCA is applied conventionally. Providing for this capacity enables individual community members and stakeholders to deliberate in defining their problem and evaluating alternative means of solving it, however, which contravenes the principle of individual sovereignty on which BCA is founded.

The pragmatism of INFFER's translation of BCA theory follows also from few regional NRM bodies having access to the skills needed for anything more than simple economic analysis (Seymour et al. 2007). Hence, the adoptability of economics-based decision tools in this context depends considerably on their ease of use. The standard procedure for applying INFFER facilitates ease of use by avoiding the need to apply specialised techniques in valuing project benefits (especially in valuing non-market benefits for which sophisticated skills are often required). With this procedure valuing project benefits in non-monetary units, however, it is not possible to calculate the NPV/I measure (nor a non-monetary proxy for it) by which BCA studies conventionally rank projects for the purposes of allocating a budget between them. Instead, the INFFER project team has formulated a Benefit: Cost Index (BCI) to be used for this purpose, without acknowledging that this criterion is a second-best proxy for the NPV/I measure. Ease of INFFER application is facilitated also by it involving 'a highly structured process, with a lot of detailed guidance provided' (Anon. 2009). This structuring necessarily injects value judgements of the INFFER project team into the process, however, which reduces, in turn, the scope to accommodate value judgements reached deliberatively in collaborative and participative arena.

The decision-making process involved in INFFER is therefore unlikely to wholly satisfy devotees of either BCA or deliberative methods of environmental valuation. However, the level of buy-in to INFFER from governments and regional NRM bodies indicates that the compromise reached between these camps is one that a significant proportion of governmental and community-based stakeholders in Australian environmental management are prepared to live with. Even so, the discussion in this chapter highlighted two weaknesses of INFFER that may limit expansion of its use. The first of these stems from the INFFER project team's decision in designing the framework to not allow users to account for benefits or costs of projects arising from their consequences for the community and other capacities needed for successful pursuit of longer-term environmental goals. The team's argument against 'investing in the community for its own sake' misses the point that pursuit of such goals is unlikely to succeed without current investment decisions in this domain accounting systematically for their impacts on the community and other capacities will be needed into the longer term.

The second possible weakness of INFFER that was highlighted in this chapter, and that may restrict its use, concerns its limitations as a vehicle for making investment choices consistent with the philosophy of integrated environmental (or catchment) management. This weakness derives from INFFER being

structured as an asset-based approach rather than a treatment-based approach. This choice followed from a view that an asset-based approach is more likely to help people focus on outcomes (although this view remains unsubstantiated by hard evidence). However, as argued by Black et al. (2002, 2004) in respect of the asset-based SIF, an asset-based method of identifying and prioritising investment opportunities tends to be biased against integrated projects that may not generate high-level benefits for any single asset, or even for a particular group of assets within the same class (e.g., wetlands), but which generate benefits across a diversity of asset classes that accumulate to a high level of benefit. Nevertheless, the INFFER team observes that their asset-based approach has been used to assess projects in which multiple disparate asset types were combined into a single asset for the purposes of assessment. They recognise, however, that a weakness can arise in such applications because (a) questions in the Project Assessment Form need to be answered for the asset as a whole rather than separately for each of the asset types comprising the combined asset, and (b) the appropriate answers may differ significantly across these asset types. They acknowledge this as a cost of simplifying INFFER sufficiently to make it useable, but argue that treatment-based approaches would need to be simplified to a similar extent to make them useable. This points to the importance of comparing what can realistically be implemented on both sides, rather than comparing a real-world method on one side with an ideal method on the other (David Pannell, pers. comm., June 2011).

Although the INFFER project team has emphasised the distinction between their method and that of multi-criteria analysis (MCA), their method is likely at times to present decision makers with cognitively taxing tasks that incorporation of elements of an MCA method might help break down into more manageable steps – while also adding transparency to the process. Moreover, the INFFER team has acknowledged that the BCI on which it focuses is not the only criterion that decision makers should consider in arriving at a final ranking of projects. MCA would provide for a structured and transparent method of balancing the BCI criterion against other criteria (e.g., equity, sustainability, employment, legitimacy) in arriving at this final ranking.

6. ECONOMIC ACCOUNTABILITY UNDER COMMUNITY-BASED ENVIRONMENTAL MANAGEMENT: CHOOSING AN APPROACH

A range of methods for evaluating decisions concerned with investment in the natural environment were reviewed in the preceding chapters. These methods were broadly categorised in chapters 2 to 4 under the headings of benefit-cost analysis (BCA), multi-criteria analysis (MCA) and deliberative methods. Within each of these categories, however, we found multiple methods of a more specific nature or purpose.

Some of these more specific methods were found to span two or more of the categories. For instance: deliberative MCA spans the categories of MCA and deliberative methods; deliberative monetary valuation spans the categories of deliberative methods and BCA; and methods have been applied also that incorporate BCA results into an overarching framework of MCA. In chapter 5, we found that although the Investment Framework for Environmental Resources (INFFER) focuses primarily on introducing a BCA mindset into the process of evaluating environmental investment decisions, it also provides considerable scope for deliberation by communities and other stakeholders in defining problems and solutions and valuing the benefits of the solutions that are identified. We found also that scope exists to incorporate INFFER output on the relative economic efficiency of projects into a broader MCA framework capable of weighing up a wider range of criteria (e.g., equity, sustainability).

Methods of evaluating decisions – including those about environmental investment – are 'value articulating institutions' (Jacobs 1997). Such institutions define the rules to be followed in the process of evaluation. Vatn (2005 pp. 301-302) identified these rules as concerned with:

- participation – who participates, on what premises (position or role), and by which method (e.g., responding to a survey, attending a meeting, written submission);
- what counts as data, and what form it should take (e.g., prices, weights, arguments); and
- the kind of data handling procedures to be used (how data is produced, weighed and aggregated).

Choosing a method to use in evaluating a particular environmental investment decision is thus a choice between alternative institutions. Economists have long concerned themselves with institutional choices in general, and the consensus now in neoclassical welfare economics is that such choices are evaluated most appropriately using a comparative institutions approach 'in which the relevant choice is between alternative real institutional arrangements' (Demsetz 1969 p. 1)[48]. It follows that no single institutional arrangement is universally optimal, but rather that the optimal arrangement in any given context depends on the particulars of the context. Hence, it is mistaken to expect any single method of economic evaluation (i.e., value articulating institution) to be optimal across all contexts where support for environmental investment decisions is required.

[48] The 'remediableness criterion' of Williamson (2000) – a leading exponent of the 'new institutional economics' which extends neoclassical economic theory beyond market choices to institutional choices – essentially restates the comparative institutions approach.

In respect of economic valuation, this mistake has been made most commonly by advocates of BCA, who tend to compare alternative evaluation methods not with BCA as it would *feasibly* be practised but with BCA as it would *ideally* be practised (see section 3.6). Although this strategy involves a 'nirvana fallacy', it serves to justify criticism of alternatives to BCA because they are imperfect. A comparative institutions approach, in contrast, would recognise that all feasible evaluation methods are imperfect and that the objective is to identify the best one given the situation at hand. As Vatn (2005 p. 360) stated succinctly, 'No VAI [value articulating institution] is ideal'.

Black et al. (2002, 2004) essentially applied a comparative institutions approach in comparing the suitability of alternative categories of evaluation method (economic/BCA, MCA and deliberative) for the Western Australian Government's Salinity Investment Framework (SIF). Each of the method categories was assessed for its performance against various criteria deemed important for decisions in natural resource management. These criteria included: (a) ability to account for macro-policy concerns (e.g., equity and macro-economic effects); (b) workability given available data and skills; and (c) community perceptions of the legitimacy of the method. These authors also assessed each of the method categories against five questions posed by the consultancy brief (including 'Does the tool include involvement of stakeholders?' and 'Is it applicable at various levels of detail (i.e., state, regional and subregional levels)?') to which they were responding. They found that 'the choice of an appropriate decision tool for the SIF is extremely complex' since no one method dominated in respect of the yardsticks applied (Black et al. 2004 p. 182).

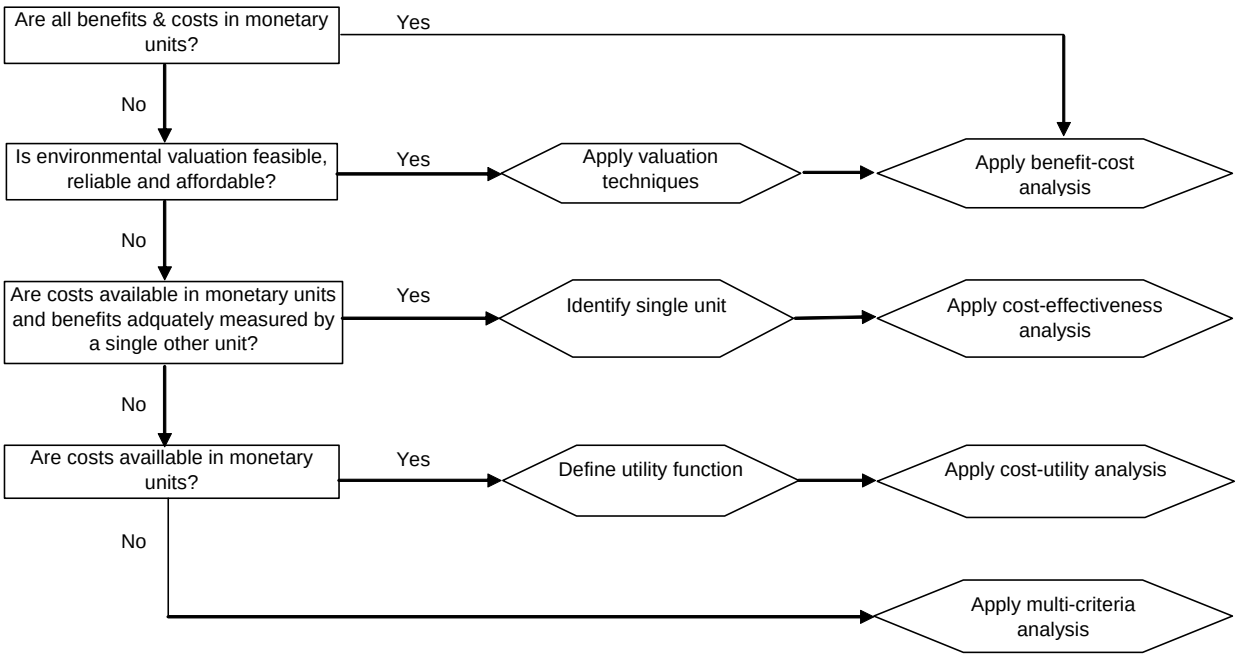
Based on their wide-ranging review of the applicability of various decision evaluation methods to the context of natural resource management, Hajkowicz et al. (2000 p. 118) similarly found it 'evident that no single NRM decision support method is clearly dominant'. Further, they admitted that '[a]t the commencement of writing this text we would have wished to fill this section with a straightforward flowchart for 'how to choose an appropriate decision support method for NRM'. Having reviewed a wide range of the methods available we feel that such a chart would be misleading' (ibid. p. 118).

Nevertheless, the lead author of that review felt able to offer such a flowchart (shown in Figure 6.1) when the range of evaluation methods was constrained to those he referred to as 'economic' (Hajkowicz 2008a). Included in this category of methods were BCA, cost-effectiveness analysis (CEA, see section 2.2.7), cost-utility analysis (CUA, see section 3.3.1), and MCA. He argued, 'None [of the four methods] is inherently better or more robust and all are based on solid theoretical foundations. The key determinant of which to use relates to valuation' (ibid. p. 3). He reasoned that:

> If benefits are adequately measured in monetary units, then BCA provides an adequate framework. If this is not the case, the analyst will need to contemplate nonmarket valuation (NMV), which will require attention to both reliability and cost-effectiveness. If it is decided that NMV is not feasible or worthwhile, then CUA may be appropriate. If there is no monetary cost data, e.g., the options are strategic policy directions, then MCA can be used (ibid. p. 3).

The proposition that the choice between these methods should be determined predominantly by the feasibility, reliability and affordability of valuing environmental non-market benefits represents a strong value judgement by this author. A wide range of factors other than issues with non-market environmental valuation were identified in chapters 2 and 3 as relevant to the choice between BCA methods (including CEA) and MCA methods (including CUA). The choice between these methods is therefore likely in many cases to be less straightforward than suggested by Hajkowicz (ibid.).

Figure 6.1: Process for choosing whether to use BCA, CEA, CUA or MCA



Source: Adapted from Figure 2 in Hajkowicz (2008a p. 3).

The discussion in previous chapters highlighted the wide range of criteria that commentators on evaluation methods for environmental decisions have stated or implied should be considered when choosing between such methods. A non-exhaustive list of such criteria is presented in Table 6.1. The comments in the table suggest that none of the BCA, MCA or deliberative types of evaluation method is likely to be scored highest across all the criteria, even by someone particularly committed to one of the method types. Hence, a choice between the methods would normally depend on the relative importance that a chooser places on each of the criteria and on how highly they rank or score each method type against each of the criteria[49]. For instance, mainstream environmental economists would presumably score BCA most highly against the first three criteria in Table 6.1, and probably also against the 'internal consistency' and 'protection against strategic manipulation' criteria. They would likely also weight these criteria more highly than the remaining 14 criteria[50].

The focus in the present project is on identifying a method for economic evaluation of environmental investment decisions that is consistent with the stated premises on which Australian governments have devolved significant powers over such investment decisions to regional and other community-based organisations. As discussed in section 1.4, this devolution has been justified by the benefits of a community-based approach for developing the capacities of landholders and other stakeholders to respond self-reliantly to the environmental challenges they face.

[49] Since the criteria presented in Table 6.1 are not mutually exclusive (e.g., 'consistency with neoclassical welfare economics' overlaps with 'alignment with an economic way of thinking'), their use in an MCA model would result in double-counting. The criteria would need to be adapted for such a use.

[50] It is not only professionals, be they economist, engineer or ecologist, that would weight the criteria differently. Those affected by consequences of a decision may also place different weights, given that different evaluation methods may tend to favour different interests.

Table 6.1: A selection of criteria for choosing between methods for evaluating environmental decisions

Criterion	Comments
Consistency with neoclassical welfare economics	An advantage claimed for BCA. However, MCA methods based on multi-attribute utility theory (MAUT) share some of the theoretical foundations of BCA. Some stakeholders may view consistency with neoclassical welfare economics as a disadvantage.
Alignment with an economic way of thinking	An advantage claimed for BCA, although MCA can be structured along the lines of an economic way of thinking.
Consistency with a focus on environmental outcomes	An advantage sometimes claimed for BCA, although each of the methods can be applied consistently with a focus on environmental outcomes. The possible advantage for BCA in this respect derives from its use of (shadow) market prices to value decision impacts, where these prices are reasoned to reflect the marginal social utility of any outcome. MCA and deliberative methods are not as constrained theoretically to focus on outcomes, so a risk exists that they may stray from this focus. This risk may be managed with structured procedures that serve to maintain an outcomes focus.
Ability to account for incommensurable values	A weakness of BCA. MAUT-based MCA methods score different values against relevant scales, but subsequently combine the scores into an overall score. Non-MAUT-based MCA methods do not combine scores into an overall score. Deliberative methods are well-suited to accounting for incommensurate values.
Scope to reflect philosophy of integrated environmental (catchment) management	In principle, each of the methods has equal scope to account for the whole range of benefits that a project may generate for different natural and environmental resources (and also in respect of other social and economic assets). In practice, however, BCA is likely to face greater problems in accounting for those benefits that cannot be readily valued using existing market prices.
Acceptability of underlying value judgements (to governments & other investors, and to communities & others whose cooperation is required)	Such value judgments may relate to (a) commensurability of different values; (b) appropriateness of measuring environmental and other intangible values on a monetary basis; (c) procedural fairness and distributive fairness (equity).
Acceptability of results (to governments & other investors, and communities & others whose cooperation is required)	Perceived validity of BCA results can be lessened by controversy concerning any non-market values incorporated in the analysis. On the other hand, governments and other stakeholders may prefer the clear-cut answers from BCA to the conditional answers often emerging from MCA or deliberative methods. (However, stakeholders can also be sceptical of clear-cut answers to problems they know are far from simple). Acceptability of results to a stakeholder may also depend on the degree to which they have participated in the process of deriving them, and thus gained trust in and ownership of the results. To the extent that deliberative and MCA methods typically allow for greater stakeholder participation than BCA, their results may find greater acceptance.
Internal consistency	An advantage claimed for BCA because it applies a single coherent body of theory in identifying and measuring all relevant values. Internal consistency in applying a method reduces risks of double counting or of introducing biases in measuring different values. However, structured approaches to MCA can strengthen internal consistency in its application.

Table 6.1 (continued)

Criterion	Comments
Protection against strategic manipulation	An advantage claimed for BCA due to (a) the rigour imposed by its reliance on a single coherent body of theory, (b) its adherence to the principle of individual sovereignty which limits inclusion of values to those of defined (sets of) individuals, and (c) its provision of clear-cut answers which provide discipline on decision makers. Even so, considerable scope for strategic manipulation remains due to reliance of BCA analysts on scientists and others for the data and assumptions they require, and also due to the 'black box' nature of BCA techniques.
Transparency	An advantage claimed for MCA (sometimes referred to as a 'glass-box' method), especially where simpler MCA algorithms are applied. BCA techniques are sometimes criticised for their 'black-box' nature, and deliberative methods have been criticised for lacking transparency in terms of clarifying all the value judgements and reasoning upon which a decision was made.
Ability to account for transaction (including political) costs	BCA often has difficulty here. MCA offers scope to include transaction costs of options as a distinct criterion.
Ability to accommodate values arising from community-based or other deliberative processes	BCA's adherence to the principle of individual sovereignty renders it unable to accommodate values arising from deliberation. In contrast, deliberatively-determined values can readily be facilitated and accounted for by deliberative methods or deliberative approaches to MCA.
Affordability	Financial costs in applying BCA can be high when consultants need to be paid, and particularly when sophisticated non-market valuation (NMV) techniques are employed. Costs of NMV may be reduced by using benefits-transfer methods where appropriate data is available, although validity concerns may arise. MCA and deliberative methods can involve significant financial costs when consultants are required and/or sophisticated techniques for applying these methods are employed. Reliance on deliberative methods (including within MCA) can also be costly in terms of time demands on participants.
Ease of use	Considerable skill is required in applying each of the types of methods – BCA, MCA and deliberative – to a standard at which confidence in the results is justified. Structured procedures may be developed that facilitate ease of application by non-experts. The skill level needed for BCA studies involving sophisticated NMV techniques is higher again, and is not easily alleviated by providing structured procedures.
Timeliness	Limited availability of appropriately-skilled practitioners can lessen the timeliness with which any of the methods is applied (e.g., in respect of funding-submission deadlines). A need to use sophisticated NMV techniques when applying BCA may especially cause timeliness issues. Reliance on deliberative methods (including within MCA) may lessen timeliness when problems arise in coordinating involvement of the relevant participants.
Facilitation of stakeholder learning	An advantage claimed for deliberative methods and MCA methods (particularly where deliberative processes are incorporated). This advantage arises from the greater participation of decision makers and other interested parties in the application of these methods, compared with the more expert-driven process involved in applying BCA.
Ability to reduce conflict and facilitate cooperation	An advantage claimed particularly for deliberative methods, but also for MCA methods that incorporate deliberative processes.

Table 6.1 (continued)

Criterion	Comments
Consistency with recognition that the decisions at issue are concerned with complex adaptive systems	BCA recognises only mechanistic relationships. Unless it forfeits its internal consistency, therefore, it cannot account for consequences of decisions arising from complex adaptive systems (e.g., related to resilience, adaptive capacity, path dependence, irreversibility). MCA and deliberative methods are not similarly constrained.
Scope to account for effects of decisions on ongoing social capacities	Social capacities (e.g., trust, reciprocity, social norms, peer pressure) typically emerge from interactions within complex adaptive systems. Unless BCA forfeits its internal consistency, therefore (see above), it is unable to account for the consequences of a decision for the social capacities needed for longer-term success in environmental management.

Part of these benefits relates to the potential advantages of a community-based approach for fostering collaboration between stakeholders that has scope to move them towards shared value systems and thus towards greater 'community ownership' of decisions that are made. A method for economic evaluation of environmental investment decisions needs to be consistent with these premises, and be capable of accounting for these kinds of benefits, if it is not to work against the prospects of realising the stated aspirations of Australian governments. This is the standard of objectivity that Bromley (1989, 2007) and Schmid (1989) have set for economists; namely, that economists take the goals set by a client (e.g., government, community-based body, etc.) as given and, with minimum subjective input, advise the client how best to achieve those goals (see section 1.5).

An economist committed to providing objective advice on what evaluation method is most consistent with stated aspirations for collaborative community-based environmental management would consider a broader range of criteria than would a mainstream environmental economist. This broader set of criteria may include all 19 criteria identified in Table 6.1, and possibly additional criteria relevant to specific contexts. This economist would need to weight the criteria 'alignment with an economic way of thinking' more highly than 'consistency with neoclassical welfare economics', and be sensitive to the expectations of her client/s when weighing up the relative importance of each of the other criteria. With a minimum of subjective input, she would need also to elicit from her clients their judgements of how each evaluation method would perform against each of the criteria.

Such an objective process of choosing the best method of economic evaluation for a given context would recognise that the choice is not only between BCA, MCA and deliberative methods as 'pure types'. To start with, each of these method types encompasses multiple variations from which a specific method must be chosen. Moreover, method types can often usefully complement one another when they are combined. Black et al. (2004 p. 166) referred to the array of method types on offer as a 'multi-purpose workbench' and noted that 'the intrinsic difficulties associated with each [evaluation] tool has encouraged a growing number of practitioners to explore the potential for combining tools'. Hajkowicz et al. (2000 p. 118) observed similarly that there is 'much value to be gained' by judiciously combining the method types to shore up each other's weaknesses.

In the final analysis, however, the scope for such an objective process to choose between economic evaluation methods depends on how 'economic' is understood by the client who desires evaluation of this kind. If the client understands this term as synonymous with the logic of neoclassical welfare economics, or views BCA as the only valid method of economic evaluation, then the scope of the process is constrained to choosing between variants of the BCA method (e.g., between different

techniques for non-market valuation). To ensure that a client's choice of BCA is well-informed, the limitations of this method in accounting for the benefits of collaborative community-based environmental management would need to be explained up-front to the client.

Many politicians, policy makers and community leaders do continue to regard BCA (together with cost-effectiveness analysis) as the only legitimate approach to economic evaluation. Few of these would be aware that the theoretical underpinnings of this approach have come under serious and sustained criticism from within the mainstream of the economics discipline (see section 1.6). Many would not be aware that alternatives to BCA for economic evaluation exist, and that growing numbers of economists are promoting and applying these alternatives. Such economists, responding to calls from their colleagues (e.g., Norgaard 1989; 2005; Davis 2006; Söderbaum 2000; Gowdy et al. 2005), have come to accept the need for a more pluralistic approach to economic evaluation. Moreover, there is a steady increase in recognition among governments and other stakeholders that methods of evaluation other than BCA can be applied consistently with an economic way of thinking, and that these methods can sometimes be more appropriate.

The Investment Framework for Environmental Resources (INFFER) is a notable recent development in these respects. This framework adheres to an economic way of thinking without imposing on decision makers the kinds of value judgments that underpin conventional applications of BCA and cost-effectiveness analysis. It borrows from MCA the idea of a scoring-based approach to valuing benefits from investments, and leaves community-based environmental organisations ample scope to employ deliberative methods when value judgements are required from them. Despite its accommodation of a more pluralistic approach to economic evaluation, this framework is finding acceptance among government agencies in Australia that traditionally have equated regarded BCA as synonymous with economic evaluation. Even so, INFFER could be modified in a number of ways (as identified in chapter five) to enhance its suitability for community-based economic evaluation of investment decisions.

REFERENCES

Ackerman, F., and Heinzerling, L. 2004. *Priceless: On Knowing the Price of Everything and the Value of Nothing*. New York: The New Press.

Adamowicz, W. L. 2004. 'What's it worth? An examination of historical trends and future directions in environmental valuation.' *Australian Journal of Agricultural and Resource Economics* 48 (3):419-443.

Adamowicz, W. L., Beckley, T., Hatton MacDonald, D., Just, L., Luckert, M., Murray, E., and Phillips, W. 1998. 'In search of forest resource values of indigenous peoples: Are nonmarket valuation techniques applicable?' *Society and Natural Resources* 11 (1):51-66.

Adamson, D. 2006. Avoiding non-market valuation via optimisation in R&D program investment. RSMG Discussion Note. Brisbane: Risk and Sustainable Management Group, School of Economics, University of Queensland. Available from http://www.johnquiggin.com/rsmg/wordpress/wp-content/uploads/2007/01/multiple%20research%20criteria.pdf

Alston, J. M., Norton, G. W., and Pardey, P. G. 1995. *Science under Scarcity: Principles and Practice for Agricultural Research Evaluation and Priority Setting*. Ithaca: Cornell University Press.

Ananda, J., and Herath, G. 2003. 'The use of Analytic Hierarchy Process to incorporate stakeholders preferences into regional forest planning.' *Forest Policy and Economics* 5 (1):13-26.

Anderies, J. M., Janssen, M. A., and Ostrom, E. 2004. 'A framework to analyze the robustness of social-ecological systems from an institutional perspective.' *Ecology and Society* 9 (1):Online: http://www.ecologyandsociety.org/vol9/iss1/art18

Anon. 2008a. Caring for our Country Business Plan 2009-2010. Canberra: Australian Government. Available from http://www.nrm.gov.au/publications/books/business-plan.html

———. 2008b. Caring for our Country: A better outcome for Australia's environment and natural resources. Canberra: Australian Government.

———. 2008c. INFFER cost effectiveness index, version 2. 4 October. Available from http://cyllene.uwa.edu.au/~dpannell/inffer.htm

———. 2009. INFFER frequently asked questions. Available from http://cyllene.uwa.edu.au/~dpannell/inffer-faqs.htm

Arrow, K. J. 1951. *Social Choice and Individual Values*. New York: John Wiley and Sons.

Arrow, K. J., and Raynaud, H. 1986. *Social Choice and Multicriterion Decision-Making*. Cambridge: MIT Press.

Arthur, W. B. 1988. 'Self-reinforcing mechanisms in economics.' In *The Economy as a Complex Evolving System*, ed. P. W. Anderson, K. J. Arrow and D. Pines. Redwood City: Addison Wesley. pp. 9-31.

———. 1989. 'Competing technologies, increasing returns, and lock-in by historical events.' *Economic Journal* 99 (394):116-131.

———. 1994. *Increasing Returns and Path Dependence in the Economy*. Ann Arbor: University of Michigan Press.

———. 1999. 'Complexity and the economy.' *Science* 284 (5411):107-109.

Australian Government. 2001. Intergovernmental agreement on a National Action Plan for Salinity and Water Quality. Canberra. Available from http://www.napswq.gov.au/publications/books/iga.html

———. 2007. *Best Practice Regulation Handbook*. Canberra: Office of Best Practice Regulation.

———. 2008. Caring for our Country: Questions and answers. Canberra. Available from http:www.nrm.gov.au/funding/cfoc-faq.html

Australian National Audit Office. 2008. Regional delivery model for the Natural Heritage Trust and the National Action Plan for Salinity and Water Quality. Audit Report No. 21 2007-08. Canberra: Australian National Audit Office.

Australian Transport Council. 2006. National guidelines for transport system management in Australia. Volume 3: Appraisal of initiatives. Canberra: Commonwealth of Australia.

Batie, S. S. 1989. 'Sustainable development: Challenges to the profession of agricultural economics.' *American Journal of Agricultural Economics* 71 (5):1083-1101.

Bennett, J. 2005. 'Australasian environmental economics: Contributions, conflicts and 'cop-outs'.' *Australian Journal of Agricultural and Resource Economics* 49 (3):243-261.

Berkes, F. 2007. 'Community-based conservation in a globalized world.' *Proceedings of the National Academy of Sciences of the USA* 104 (39):15188-15193.

Besley, T. 2007. 'The new political economy.' *The Economic Journal* 117:F540-F557.

Black, J. 2008. Priority setting and dryland salinity: A case for policy learning. Paper presented to the 12th Annual Conference of the International Research Society for Public Management 26-28 March, Queensland University of Technology, Brisbane.

Black, J., and Burton, M. 2002. A review of economic, multiple criteria and deliberative tools and their applicability to the Salinity Investment Framework. Report to the Investment Steering Committee for the WA Salinity Investment Framework. Perth: School of Agricultural and Resource Economics, University of Western Australia.

———. 2004. 'A decision support tool for the WA Salinity Investment Framework.' In *Dryland Salinity: Economic Issues at Farm, Catchment and Policy Levels*, ed. T. W. Graham, D. J. Pannell and B. White. Perth: Cooperative Research Centre for Plant-Based Management of Dryland Salinity, University of Western Australia.

Blamey, R. K., McCarthy, P., and Smith, R. 2000. Citizens' juries and small-group decision-making. Research Report No. 2, Citizens' Jury Project. Canberra: Research School of the Social Sciences, Australian National University. October.

Border Rivers - Gwydir Catchment Management Authority. 2006. Catchment Action Plan. Inverell.

Boulding, K. E. 1970. *Economics as a Science*. New York: McGraw-Hill.

Bouysou, D. 1990. 'Building criteria: A prerequisite for MCDA.' In *Readings in Multiple Criteria Decision Aid*, ed. C. A. Bana e Costa. Berlin: Springer Verlag. pp.

Brock, W. A., and Carpenter, S. R. 2007. 'Panaceas and diversification of environmental policy.' *Proceedings of the National Academy of Sciences of the USA* 104 (39):15206-15211.

Bromley, D. W. 1989. *Economic Interests and Institutions: The Conceptual Foundations of Public Policy*. New York: Basil Blackwell.

———. 2007. 'Environmental regulations and the problem of sustainability: Moving beyond 'market failure'.' *Ecological Economics* (63):676-683.

Buchanan, J. M. 1954. 'Social choice, democracy, and free markets.' *Journal of Political Economy* 62:114-123.

Buchanan, J. M., and Tulloch, G. 1962. *The Calculus of Consent. The Logical Foundations of Constitutional Democracy*. Ann Arbor, USA: University of Michigan Press.

Bureau of Transport Economics. 1999. *Facts and Furphies in Benefit-Cost Analysis: Transport*. Canberra: Bureau of Transport Economics.

Carr, A. 2002. *Grass Roots and Green Tape: Principles and Practices of Environmental Stewardship*. Sydney: The Federation Press.

Chakravarty, S. 1987. 'Cost-benefit analysis.' In *The New Palgrave Dictionary of Economics, vol. 1*. London. pp. pp. 687-690.

Challen, R. 2000. *Institutions, Transaction Costs and Environmental Policy: Institutional Reform for Water Resources*. Cheltenham: Edward Elgar.

Chambers, S. 2003. 'Deliberative democratic theory.' *Annual Review of Political Science* 6:307-326.

Charnes, A., Cooper, M. W., and Ferguson, R. 1955. 'Optimal estimation of executive compensation by linear programming.' *Management Science* 1:138-151.

Cleland, J. 2008. Western Australia's Salinity Investment Framework: A study of priority setting in policy and practice. Ph.D. thesis (draft), School of Agricultural and Resource Economics, University of Western Australia, Perth.

Cohen, J. 1998. 'Democracy and liberty.' In *Deliberative Democracy*, ed. J. Elster. Cambridge: Cambridge University Press. pp. 185-231.

Colander, D. 2000. 'New millennium economics: How did it get this way, and what way is it?' *Journal of Economic Perspectives* 14 (1):121-132.

Coleman, J. S. 1990. *Foundations of Social Theory*. Cambridge: Harvard University Press.

Commonwealth Grants Commission. 2004. Annual Report 2003-04. Canberra: Commonwealth Grants Commission.

Commonwealth of Australia. 2008a. Caring for our Country Business Plan 2009-2010. Canberra. Available from http://www.nrm.gov.au/publications/books/pubs/business-plan.pdf

———. 2008b. Strengthening Rural and Regional Australia. Budget statement by the Honourable Anthony Albanese, MP, the Honourable Tony Burke MP and the Honourable Gary Gray MP. Canberra: Attorney-General's Department. 13 May.

Costanza, R. 2000. 'Visions of alternative (unpredictable) futures and their use in policy analysis.' *Conservation Ecology* 4 (1):5

Crean, J. 2003. Agri-environmental conservation: The case for an environmental levy. Paper presented to the 47th Annual Conference of the Australian Agricultural and Resource Economics Society, 12-14 February, Fremantle.

Cullen, R., Fairburn, G., and Hughey, K. 2001. 'Measuring the productivity of threatened species programs.' *Ecological Economics* 39 (1):53-66.

Curtis, A., Lucas, D., Nurse, M., and Skeen, M. 2008. Achieving NRM outcomes through voluntary action: Lessons from Landcare. A discussion paper. Melbourne: Department of Sustainability and Environment.

Davis, J. B. 2006. 'The turn in economics: Neoclassical dominance to mainstream pluralism?' *Journal of Institutional Economics* 2 (1):1-20.

De Marchi, B., and Ravetz, J. R. 2001. 'Participatory approaches to environmental policy.' In *EVE Policy Brief*, ed. C. L. Spash and C. Carter. Cambridge: Cambridge Research for the Environment. pp. 18.

Demsetz, H. 1969. 'Information and efficiency: Another viewpoint.' *Journal of Law and Economics* 12 (April):1-22.

Department of Environment. 2003. Salinity Investment Framework Interim Report - Phase 1. Salinity and Land Use Impacts Series, Report no. SLUI 32. Perth: Department of Environment.

Department of Finance. 1991. *Introduction to Cost-Benefit Analysis*. Canberra: Commonwealth of Australia.

Dixit, A. K. 1996. *The Making of Economic Policy: A Transaction-Cost Politics Perspective*. Cambridge: MIT Press.

Drummond, M., O'Brien, B., Stoddart, G., and Torrance, G. 1997. *Methods for the Economic Evaluation of Healthcare Programmes*. Oxford: Oxford Medical Publications.

Dryzek, J. S. 1990. *Discursive Democracy: Politics, Policy and Political Science*. Cambridge: Cambridge University Press.

———. 2002. *Deliberative Democracy and Beyond: Liberals, Critics, Contestations*. Paperback edition ed. Oxford: Oxford University Press.

Dumsday, R. G. 2001. Policy analysis for land and water resources. Scoping project for the National Land and Water Resources Audit. Canberra: Commonwealth of Australia.

Dziegielewska, D., Tietenberg, T., and Niggol Seo, S. 2007. 'Total economic value.' In *Encyclopedia of Earth*, ed. C. J. Cleveland. Washington, D.C.: Environmental Information Coalition, National Council for Science and the Environment.

Edwards-Jones, G., Davies, B., and Hussain, S. 2000. *Ecological Economics: An Introduction*. Oxford: Blackwell Science.

Edwards, G., and Byron, N. 2001. Land degradation and rehabilitation: A policy framework. the 4th Symposium of the Australian Agricultural and Resource Economics Society: 'Public Funding of Environmental Issues', 5 October, Melbourne.

Edwards, V. M., and Steins, N. A. 1999. 'Special issue introduction: The importance of context in common pool resource research.' *Journal of Environmental Policy and Planning* 1:195-204.

Ezrahi, Y. 1990. *The Descent of Icarus: Science and the Transformation of Contemporary Democracy*. Cambridge, Massachusetts: Harvard University Press.

Falconer, K., Dupraz, P., and Whitby, M. 2001. 'An investigation of policy administrative costs using panel data for the English Environmentally Sensitive Areas.' *Journal of Agricultural Economics* 52 (1):83-103.

Farquharson, B., Hill, C., Bennett, J., and Tracey, J. 2007. Environmental economics and valuation: Towards a practical investment framework for Catchment Management Authorities in New

South Wales. Paper presented to the 51st Annual Conference of the Australian Agricultural and Resource Economics Society, Queenstown, New Zealand.

Fernandes, L., Ridgely, M. A., and van't Hof, T. 1999. 'Multiple criteria analysis integrates economic, ecological and social objectives for coral reef managers.' *Coral Reefs* 18:393-402.

Figueira, J., Salvatore, G., and Ehrgott, M., eds. 2005. *Multiple Criteria Decision Analysis: State of the Art Surveys*. New York: Springer.

Fishkin, J. S. 1997. *The Voice of the People*. New Haven, CT: Yale University Press.

Friedman, M. 1953. *Essays in Positive Economics*. Chicago: Chicago University Press.

Funtowicz, S. O., and Ravetz, J. R. 1990. *Uncertainty and Quality in Science for Policy*. Dordrecht, The Netherlands: Kluwer Academic Publishers.

Furubotn, E. G., and Richter, R. 1992. 'The New Institutional Economics: An Assessment.' In *New Institutional Economics*, ed. E. G. Furubotn. London: Edward Elgar. pp. 1-32.

Gallagher, R., and Appenzeller, T. 1999. 'Beyond reductionism.' *Science* 284 (5411):79.

Goldenfield, N., and Kadanoff, L. P. 1999. 'Simple lessons from complexity.' *Science* 284 (5411):87-89.

Government of South Australia. 1999. Application of environmental valuation in South Australia. Adelaide: Department of Environment, Heritage and Aboriginal Affairs.

Government of Victoria. 2007. *Victorian Guide to Regulation*. 2nd ed. Melbourne: Department of Treasury and Finance.

Government of Western Australia. 2000. The Salinity Strategy: Natural resource management in Western Australia. Perth.

———. 2003. Preliminary agency statement of natural resource management priorities in Western Australia. Perth: Department of Agriculture.

Gowdy, J. M., and Erickson, J. D. 2005. 'The approach of ecological economics.' *Cambridge Journal of Economics* 29:207-222.

Gowdy, J. M., and Howarth, R. B. 2007. 'Sustainability and benefit-cost analysis: Theoretical assessments and policy options.' *Ecological Economics* 63:637-638.

Gowdy, J. M., and Mayumi, K. 2001. 'Reformulating the foundations of consumer choice theory and environmental evaluation.' *Ecological Economics* 39:223-237.

Habermas, J. 1984. *The Theory of Communicative Action. Volume 1: Reason and the Rationalization of Society*. Boston, MA: Beacon Press.

Hajkowicz, S. 2007a. 'Allocating scarce financial resources across regions for environmental management in Queensland, Australia.' *Ecological Economics* 61:208-216.

———. 2007b. 'A comparison of multiple criteria analysis and unaided appproaches to environmental decision making.' *Environmental Science and Policy* 10 (3):177-184.

Hajkowicz, S., and McDonald, G. 2006a. 'The assets, threats and solvability (ATS) model for setting environmental priorities.' *Journal of Environmental Policy and Planning* 8 (1):87-102.

Hajkowicz, S. A. 2002. Regional priority setting in Queensland: A multi-criteria evaluation framework. A report to the Queensland Department of Natural Resources and Mines. Canberra: CSIRO Division of Land and Water.

———. 2008a. 'Rethinking the economist's evaluation toolkit in light of sustainability policy.' *Sustainability: Science, Practice and Policy* 4 (1):1-8.

———. 2008b. 'Supporting multi-stakeholder environmental decisions.' *Journal of Environmental Management* 88 (4):607-614.

———. 2009. 'The evolution of Australia's natural resource management programs: Towards improved targeting and evaluation of investments.' *Land Use Policy* 26 (2):471-478.

Hajkowicz, S. A., and McDonald, G. 2006b. 'The assets, threats and solvability (ATS) model for setting environmental priorities.' *Journal of Environmental Policy and Planning* 8 (1):87-102.

Hajkowicz, S. A., Young, M., Wheeler, S., McDonald, D. H., and Young, D. 2000. Supporting Decisions: Understanding Natural Resource Management Techniques. Adelaide: Policy and Economic Research Unit, CSIRO Land and Water. June.

Hanley, N. 2001. 'Cost-benefit analysis and environmental policymaking.' *Environment and Planning C: Government and Policy* 19:103-118.

Holland, A. 1997. 'The foundations of environmental decision-making.' *International Journal of Environment and Pollution* 7 (4):483-496.

Holling, C. S., Berkes, F., and Folke, C. 1998. 'Science, sustainability and resource management.' In *Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience*, ed. F. Berkes and C. Folke. Cambridge: Cambridge University Press. pp. 342-362.

Howarth, R. B., and Wilson, M. A. 2006. 'A theoretical approach to deliberative valuation: Aggregation by mutual consent.' *Land Economics* 82 (1):1-16.

Industry Commission. 1998. A full repairing lease: Inquiry into ecologically sustainable land management. Canberra: Industry Commission.

Jacobs, M. 1997. 'Environmental valuation, deliberative democracy and public decision-making.' In *Valuing Nature? Economics, Ethics and Environment*, ed. J. Foster. London: Routledge. pp. 211-231.

Janssen, M. A., and Anderies, J. M. 2007. 'Robustness tradeoffs in social-ecological systems.' *International Journal of the Commons* 1 (1):43-65.

Janssen, R. 1992. *Multiobjective Decision Support for Environmental Management*. Dordrecht: Kluwer.

———. 2001. 'On the use of multi-criteria analysis in environmental impact assessment in The Netherlands.' *Journal of Multi-Criteria Decision Analysis* 10:101-109.

Joseph, L. N., Malony, R. F., and Possingham, H. P. 2008. 'Optimal allocation of resources among threatened species: A Project Prioritization Protocol.' *Conservation Biology* 23 (2):328-338.

Joubert, A. R., Leiman, A., de Klerk, H. M., Jatua, S., and Aggenbach, J. C. 1997. 'Fynbos (fine bush) vegetation and the supply of water: A comparison of multi-criteria decision analysis and cost-benefit analysis.' *Ecological Economics* 22:123-140.

Kahn, A. E. 1966. 'The tyranny of small decisions: Market failures, imperfections, and the limits of economics.' *Kyklos* 19:23-47.

Kallis, G., Videira, N., Antunes, P., Guimarães Pereira, Â., Spash, C. L., Coccosis, H., Corral Quintana, S., del Moral, L., Hatzilacou, D., Lobo, G., Mexa, A., Paneque, P., Pedregal Mateos,

B., and Santos, R. 2006. 'Participatory methods for water resources planning.' *Environment and Planning C: Government and Policy* 24:215-234.

Keeney, R. L., and Raiffa, H. 1976. *Decisions with Multiple Objectives: Preferences and Value Tradeoffs*. New York: John Wiley and Sons.

Knight, F. 1947. *Freedom and Reform: Essays in Economic and Social Philosophy*. New York: Harper.

Lane, M. B. 2003. 'Decentralization or privatization of environmental governance? Forest conflict and bioregional assessment in Australia.' *Journal of Rural Studies* 19 (3):283-294.

Lienhoop, N., and Macmillan, D. C. 2007. 'Valuing a complex environmental change: Assessing participant performance in deliberative group-based approaches and in-person interviews for contingent valuation.' *Environmental Values* 16 (2):209-232.

Linares, P., and Romero, C. 2002. 'Aggregation of preferences in an environmental economics context: A goal programming approach.' *Omega* 30:89-95.

Macmillan, D. C., Hanley, N., and Lienhoop, N. 2006. 'Contingent valuation: Environmental polling or preference engine?' *Ecological Economics* 60 (1):299-307.

Marchamalo, M., and Romero, C. 2007. 'Participatory decision-making in land use planning: An application in Costa Rica.' *Ecological Economics* 63:740-748.

Marsh, S. P., Pannell, D. J., Seymour, E., Ridley, A., and Wilkinson, R. 2008. What's stopping regional catchment organisations from investing effectively in natural resource management? Paper presented to the 52nd Annual Conference of the Australian Agricultural and Resource Economics Society, 5-8 February, Canberra.

Marshall, G. R. 2003. Towards a resource economics for adaptive managers. Paper presented to the 47th Annual Conference of the Australian Agricultural and Resource Economics Society, 12-14 February, Fremantle. Available from http://www.ruralfutures.une.edu.au/downloads/adaptivemanagers_120.pdf

———. 2005. *Economics for Collaborative Environmental Management: Renegotiating the Commons*. London: Earthscan.

———. 2008a. Community-based, regional delivery of natural resource management: Building system-wide capacities to motivate voluntary farmer adoption of conservation practices. Canberra Rural Industries Research and Development Corporation. Available from https://rirdc.infoservices.com.au/items/08-175

———. 2008b. Investment planning under regional NRM delivery: Current processes and issues in three NSW regions. Working Paper 1 from the project 'Improving economic accountability when using decentralised, collaborative approaches to environmental decisions'. Armidale: Institute for Rural Futures, University of New England. Available from http://www.ruralfutures.une.edu.au/downloads/IE_Scoping.pdf

———. 2008c. 'Nesting, subsidiarity, and community-based environmental governance beyond the local level.' *International Journal of the Commons* 2 (1):75-97. Online: http://www.thecommonsjournal.org/index.php/ijc/article/viewFile/50/19

———. 2009. 'Polycentricity, reciprocity, and farmer adoption of conservation practices under community-based governance.' *Ecological Economics* 68 (5):1507-1520.

———. 2010a. Economic evaluation of investments in natural assets under community-based environmental governance: Developing and testing a method. Working paper 3 from the project

'Improving economic accountability when using decentralised, collaborative approaches to environmental decisions'. Armidale: Institute for Rural Futures, University of New England. Available from http://www.ruralfutures.une.edu.au/downloads/WP3.pdf

———. 2010b. Extending INFFER to account for community and other capacity spillovers in economic evaluation on environmental investments. Paper presented to the 54th Annual Conference of the Australian Agricultural and Resource Economics Society, Adelaide. Available from http://www.ruralfutures.une.edu.au/downloads/AARESMarshall2010_399.pdf

———. 2011. Improving economic accountability of investment decisions under community-based environmental governance: Final report from the project *Improving economic accountability when using decentralised, collaborative approaches to environmental decisions*. Armidale: Institute for Rural Futures, University of New England. Available from http://www.ruralfutures.une.edu.au/downloads/CERF/CERF_Final_Report.pdf

Marshall, G. R., Wall, L. M., and Jones, R. E. 1993. The role of economists in land and water management. Paper presented to the 37th Annual Conference of the Australian Agricultural Economics Society, 9-11 February, University of Sydney.

———. 1996. 'Economics of integrated catchment management.' *Review of Marketing and Agricultural Economics* 64 (2):166-176.

McAllister, D. M. 1980. *Evaluation in Environmental Planning: Assessing Environmental, Social, Economic and Political Trade-Offs*. Cambridge: MIT Press.

McCann, L., Colby, B., Easter, K. W., Kasterine, A., and Kuperan, K. V. 2005. 'Transaction cost measurement for evaluating environmental policies.' *Ecological Economics* 52 (4):527-542.

McDonald, G. T., and Hundloe, T. J. 1993. 'Policies for a sustainable future.' In *Land Degradation Processes in Australia*, ed. G. H. McTainsh and W. C. Boughton. Melbourne: Longman Cheshire. pp. 345-385.

Meppem, T., and Bourke, S. 1999. 'Different ways of knowing: A communicative turn toward sustainability.' *Ecological Economics* 30 (3):389-404.

Messner, F. 2006. 'Applying participatory multicriteria methods to river basin management: Improving the implementation of the Water Framework Directive.' *Environment and Planning C: Government and Policy* 24:159-167.

Messner, F., Zwirner, O., and Karkuschke, M. 2006. 'Participation in multicriteria decision support for the resolution of a water allocation problem in the Spree River Basin.' *Land Use Policy* 23:63-75.

Mishan, E. J., and Quah, E. 2007. *Cost Benefit Analysis*. 5th ed. London: Routledge.

Mooney, G., Irwig, L., and Leeder, S. 1997. 'Priority setting in health care: Unburdening from the burden of disease.' *Australian and New Zealand Journal of Public Health* 27 (7):680-681.

Munda, G. 1995. *Multicriteria Evaluation in a Fuzzy Environment: Theory and Applications*. Heidelberg: Physika.

Murray-Darling Basin Commission. 1996. *Cost-Sharing for On-Ground Works*. Canberra: Murray-Darling Basin Commission.

Myšiak, J. 2006. 'Consistency of results of different MCA methods: A critical review.' *Environment and Planning C: Government and Policy* 24:257-277.

Nancarrow, B. E., Johnston, C. S., and Syme, G. J. 2002. *Community Perceptions of Roles, Responsibilities and Funding for Natural Resource Management in the Moore Catchment*. Perth: Australian Research Centre for Water in Society, CSIRO.

Natural Resource Management Ministerial Council. 2002. National natural resource management capacity building framework. Canberra: NRM Ministerial Council. 25 July.

Natural Resources Commission. 2005. Standard for Quality Natural Resource Management. Document No. D05/5274. Sydney.

———. 2008a. Allocating NRM funding between NSW Catchment Management Authorities. Sydney: Natural Resources Commission.

———. 2008b. Progress report on effective implementation of Catchment Action Plans. Sydney: Natural Resources Commission.

Neef, A. 2009. 'Transforming rural water governance: Towards deliberative and polycentric models?' *Water Alternatives* 2 (1):53-60.

Nelson, R., Howden, M., and Stafford Smith, M. 2008. 'Using adaptive governance to rethink the way science supports Australian drought policy.' *Environmental Science and Policy* 11:588-601.

Nelson, R. H. 1987. 'The economics profession and the making of public policy.' *Journal of Economic Literature* 25:49-91.

Nicholson, C. 2006. 'Guest editorial.' *SALT Magazine* 15:2.

Niemeyer, S., and Spash, C. L. 2001. 'Environmental valuation analysis, public deliberation and their pragmatic syntheses: A critical appraisal.' *Environment and Planning C: Government and Policy* 19 (4):567-586.

Nijkamp, P., Rietveld, P., and Voogd, H. 1990. *Multicriteria Evaluation in Physical Planning*. Amsterdam: North Holland.

Nijkamp, P., and van Delft, A. 1977. *Multi-Criteria Analysis and Regional Decision-Making*. Leiden: Nijhoff.

Norgaard, R. B. 1989. 'The case for methodological pluralism.' *Ecological Economics* 1:37-57.

Norgaard, R. B., and Baer, P. 2005. 'Collectively seeing complex systems: The nature of the problem.' *Bioscience* 55 (11):953-960.

North, D. C. 1990. *Institutions, Institutional Change and Economic Performance*. Cambridge: Cambridge University Press.

Norton, B., Costanza, R., and Bishop, R. C. 1998. 'The evolution of preferences: Why 'sovereign' preferences may not lead to sustainable policies and what to do about it.' *Ecological Economics* 24:193-211.

Norton, B. G. 2005. *Sustainability: A Philosophy of Adaptive Ecosystem Management*. Chicago: University of Chicago Press.

Norton, B. G., and Noonan, D. 2007. 'Ecology and valuation: Big changes needed.' *Ecological Economics* 63:664-675.

NSW Government. 2006. State Plan: A new direction for NSW. Sydney: Premier's Department.

O'Neill, J. F. 2001. 'Representing people, representing nature, representing the world.' *Environment and Planning C: Government and Policy* 9 (4):483-500.

O'Sullivan, A., and Sheffrin, S. M. 2003 *Economics: Principles in Action*. Upper Saddle River, New Jersey Pearson Prentice Hall.

Odum, W. E. 1982. 'Environmental degradation and the tyranny of small decisions.' *Bioscience* 32 (9):728-729.

Olson, M. 1965. *The Logic of Collective Action*. Cambridge: Harvard University Press.

———. 1982. *The Rise and Decline of Nations*. New Haven: Yale University Press.

Ostrom, E. 1999. 'Coping with tragedies of the commons.' *Annual Review of Political Science* 2:493-535.

———. 2005. *Understanding Institutional Diversity*. Princeton: Princeton University Press.

Ostrom, E., Gardner, R., and Walker, J. 1994. 'Cooperation and social capital.' In *Rules, Games and Common-Pool Resources*, ed. E. Ostrom, R. Gardner and J. Walker. Ann Arbor: University of Michigan Press. pp. 319-329.

Ostrom, V., Tiebout, C. M., and Warren, R. 1999[1961]. 'The organization of government in metropolitan areas: A theoretical inquiry.' In *Polycentricity and Local Public Economies: Readings from the Workshop in Political Theory and Policy Analysis*, ed. M. D. McGinnis. Ann Arbor: University of Michigan Press. pp. 31-51.

Pagan, P. G. 2003. Laws, customs and rules: Identifying the characteristics of successful water management institutions. Paper presented to the Conference on Institutional Issues in Water Resource Allocation: Lessons from Australia and Implications for India (Convened for ACIAR Project ADP2001/014: Improving Water Resource Management in India's Agricultural Search for Effective Institutional Arrangements and Policy Frameworks), 17-18 July 2003, Beechworth, Victoria.

Paneque Salgado, P., Corral Quintana, S., Guimarães Pereira, A., and del Moral Ituarte, L. 2009. 'Participative multi-criteria analysis for the evaluation of water governance alternatives: A case in the Costa del Sol (Málaga).' *Ecological Economics* 68:990-1005.

Pannell, D., and Roberts, A. 2007. Balancing investment in localised and dispersed NRM assets: Centre for Environmental Economics and Policy, University of Western Australia. Available from http://www.ceep.uwa.edu.au/publications/balancing-investment-nrm

Pannell, D. J. 1996. 'Compulsory use of benefit-cost analysis for research proposals is counter-productive.' *REGAE News* 7:2-4.

———. 1997. 'Compulsory use of benefit-cost analysis for research proposals: Reply.' *REGAE News* 11:2-4.

———. 2001. 'Dryland salinity: Economic, scientific, social and policy dimensions.' *Australian Journal of Agricultural and Resource Economics* 45 (4):517-546.

———. 2007. Comparison of SIF1, SIF2 and SIF3. Available at http://cyllene.uwa.edu.au/~dpannell/sif123.htm

———. 2008a. 'Public benefits, private benefits, and policy interventions for land-use change for environmental benefits.' *Land Economics* 84 (2):225-240.

———. 2008b. The public: private benefits framework. Pannell Discussions no. 129. Available from http://cyllene.uwa.edu.au/~dpannell/pd/pd0129.htm

Pannell, D. J., Lefroy, T., and McFarlane, D. J. 2000. A framework for prioritising Government investment in salinity in Western Australia. Report to the SIF Working Group. Perth: State Salinity Council.

Pannell, D. J., and Ridley, A. M. 2008. Lessons from dryland salinity policy experience in Australia. the 2nd International Salinity Forum, 31 March - 3 April, Adelaide.

Pannell, D. J., Roberts, A. M., Park, G., Curatolo, A., Marsh, S., and Alexander, J. 2009a. INFFER: Investment Framework For Environmental Resources. INFFER Working Paper 0901. Perth: University of Western Australia. Available from http://cyllene.uwa.edu.au/~dpannell/dp0901.htm

Pannell, P., Roberts, A., Park, G., Alexander, J., Curatolo, A., Spry, S., and Marsh, S. P. 2009b. INFFER Project Assessment Form (PAF) Instruction Manual. Online: http://cyllene.uwa.edu.au/~dpannell/inffer.htm

Park, G., Pannell, D., Curatolo, A., Roberts, A., Spry, S., and Marsh, S. 2011. Summary of INFFER steps. Available from http://cyllene.uwa.edu.au/~dpannell/inffer-steps.htm (accessed 30 May 2011).

Peacock, A. 1973. 'Cost-benefit analysis and the political control of public investment.' In *Cost-Benefit and Effectiveness: Studies and Analysis*, ed. J. N. Wolfe. London: Allen and Unwin. pp. 17-29.

Perry, C., and Dillon, J. L. 1978. 'Multiple objectives and uncertainty in ex-ante project evaluation.' In *Australian Project Evaluation: Selected Readings*, ed. J. C. McMaster and G. R. Webb. Sydney.: Australia and New Zealand Book Company.

Pezzey, J. C. V., and Toman, M. A. 2002. 'Progress and problems in the economics of sustainability.' In *The International Yearbook of Environmental and Resource Economics 2002/2003*, ed. T. Tietenberg and H. Folmer. Cheltenham: Edward Elgar. pp. 165-232.

Porter, T. M. 1995. *Trust in Numbers: The Pursuit of Objectivity in Science and Public Life*. Princeton, New Jersey: Princeton University Press.

Prato, T. 1999. 'Multi attribute decision analysis for ecosystem management.' *Ecological Economics* 30:207-222.

Prato, T., and Herath, G. 2007. 'Multiple-criteria decision analysis for integrated catchment management.' *Ecological Economics* 63:627-632.

Proctor, W. 2001. Valuing Australia's ecosystem services using a deliberative multi-criteria approach. Paper presented to the Fourth International Conference of the European Society for Ecological Economics, 3-7 July, Cambridge, England.

Proctor, W., and Drechsler, M. 2006. 'Deliberative multicriteria evaluation.' *Environment and Planning C: Government and Policy* 24 (2):169-190.

Putnam, R. D. 1993. *Making Democracy Work: Civic Traditions in Modern Italy*. Princeton: Princeton University Press.

Randall, A. 1999. 'A new look at the old problem of externalities.' *Choices* First Quarter:29-32.

Refsgaard, K. 2006. 'Process-guided multicriteria analysis in wastewater planning.' *Environment and Planning C: Government and Policy* 24:191-213.

Renn, O., Webler., T., Rakel, H., Dienel, P., and Johnson, B. 1993. 'Public participation in decision-making: A three-step procedure.' *Policy Science* 26:189-214.

Resource Assessment Commission. 1992. *Multi-Criteria Analysis as a Resource Assessment Tool. RAC Research Paper No. 6*. Canberra.

Ridley, A. M., and Pannell, D. J. 2007. SIF3: An investment framework for managing dryland salinity in Australia, version 42. SIF3 Working Paper 0705. Perth: Future Farm Industries CRC. Available from http://cyllene.uwa.edu.au/~dpannell/sif3-42.pdf

———. 2008. Piloting a systematic framework (SIF3) for public investment in regional natural resource management in dryland salinity in Australia. Paper presented to the 2nd International Salinity Forum: Salinity, Water and Society – Global Issues, Local Action, 31 March – 3 April, Adelaide

Rippe, K. P., and Schaber, P. 1999. 'Democracy and environmental decision-making.' *Environmental Values* 8:75-88.

Rittel, H., and Webber, M. M. 1973. 'Dilemmas in the general theory of planning.' *Policy Sciences* 4:155-169.

Roy, B. 1996. *Multicriteria Methodology for Decision Aiding*. Dordrecht: Kluwer.

Sagoff, M. 1998. 'Aggregation and deliberation in valuing environmental public goods: A look beyond contingent pricing.' *Ecological Economics* 24 (2-3):213-230.

Samuels, W. J. 1989. 'Foreword.' In *Benefit-cost Analysis: A Political Economy Approach*, ed. A. A. Schmid. Boulder: Westview Press. pp. xiii-xviii.

Schmid, A. A. 1989. *Benefit-cost Analysis: A Political Economy Approach*. Boulder: Westview Press.

———. 2000. 'Affinity as social capital: Its role in development.' *Journal of Socio-Economics* 29 (2):159-171.

———. 2004. *Conflict and Cooperation: Institutional and Behavioral Economics*. Malden: Blackwell Publishing.

Sen, A. K. 1995. 'Rationality and social choice.' *American Economic Review* 85 (1):1-24.

Seymour, E., Pannell, D. J., Ridley, A., Marsh, S. P., and Wilkinson, R. 2007. Capacity needs for technical analysis and decision making within Australian catchment management organisations. Report for CVCB project no. UWA-92A. Perth: Cooperative Venture for Capacity Building. July.

Shumway, C. R. 1981. 'Subjectivity in *ex ante* research evaluation.' *American Journal of Agricultural Economics* 63 (1):169-173.

Sinden, J. A., Downey, P. O., Hester, S. M., and Cacho, O. 2008. Valuing the biodiversity gains from protecting native plant communities from bitou bush (*Chryanthemoides monilifera* subsp. *rotundata* (DC.) T. Norl.) in New South Wales: Application of the defensive expenditure method. Paper presented to the 52nd Annual Conference of the Australian Agricultural and Resource Economics Society, February, Canberra.

Sinden, J. A., and Griffith, G. 2007. 'Combining economic and ecological arguments to value the environmental gains from control of 35 weeds in Australia.' *Ecological Economics* 61:396-408.

Sinden, J. A., and Thampapillai, D. J. 1995. *Introduction to Benefit-Cost Analysis*. Melbourne: Longman Australia.

Söderbaum, P. 2000. *Ecological Economics: A Political Economics Approach to Environment and Development*. London: Earthscan.

———. 2008. *Understanding Sustainability Economics: Towards Pluralism in Economics*. London: Earthscan.

Spash, C. L. 2001. 'Broadening democracy in environmental policy processes.' *Environment and Planning C: Government and Policy* 19:475-481.

———. 2002. 'Informing and forming preferences in environmental valuation: Coral reef biodiversity.' *Journal of Economic Psychology* 23 (5):665-687.

———. 2007. 'Deliberative monetary valuation (DMV): Issues in combining economic and political processes to value environmental change.' *Ecological Economics* 63:690-699.

———. 2008. 'How much is that ecosystem in the window? The one with the bio-diverse trail.' *Environmental Values* 17:259-284.

Spash, C. L., and Vatn, A. 2006. 'Transferring environmental value estimates: Issues and alternatives.' *Ecological Economics* 60:379-388.

Stirling, A. 1997. 'Multi-criteria mapping: Mitigating the problems of environmental valuation?' In *Valuing Nature: Economics, Ethics and Environment*, ed. J. Foster. London: Routledge. pp. 186-210.

Strijker, D., Sijtsma, F. J., and Wiersma, D. 2000. 'Evaluation of nature conservation: An application to the Dutch Ecological Network.' *Environmental and Resource Economics* 16:363-378.

Sugden, R. 2005. Integrating cost-benefit analysis and multi-criteria analysis of flood and coastal erosion risk management projects. R&D Project Record FD2018/PR2.

Sugden, R., and Williams, A. 1978. *The Principles of Practical Cost-Benefit Analysis*. Oxford, UK: Oxford University Press.

Thorpe, J. 2008. Assessing policy instruments: A report for the Department of the Environment, Water, Heritage and the Arts: PriceWaterhouseCoopers.

Toyne, P., and Farley, R. 2000. The Decade of Landcare: Looking backward – looking forward. Discussion Paper No. 30. Canberra: The Australia Institute, Australian National University. July.

URS. 2003. Non-market valuation and holistic assessment: Part 1. Non-market benefits of research and development. Final report prepared for Land and Water Australia. Canberra: URS.

van Bueren, M., and Bennett, J. 2004. 'Towards the development of a transferable set of value estimates for environmental attibutes.' *Australian Journal of Agricultural and Resource Economics* 48 (1):1-32.

van Pelt, M., Kuyvenhoven, A., and Nijkamp, P. 1990. Project appraisal and sustainability: The applicability of cost-benefit and multi-criteria analysis. Wageningen Economc Papers 5. Wageningen: Wageningen Agricultural University.

Vatn, A. 2005. *Institutions and the Environment*. Cheltenham: Edward Elgar.

Vatn, A., and Bromley, D. W. 1994. 'Choices without prices without apology.' *Journal of Environmental Economics and Management* 26 (2):129-148.

Venn, T. J., and Quiggin, J. 2007. 'Accommodating indigenous cultural heritage values in resource assessment: Cape York Peninsula and the Murray-Darling Basin, Australia.' *Ecological Economics* 61:334-344.

von Neumann, J., and Morgenstern, O. 1944. *Theory of Games and Economic Behavior*. Princeton: Princeton University Press.

Walker, B., and Salt, D. 2006. *Resilience Thinking: Sustaining Ecosystems and People in a Changing World*. Washington, D.C.: Island Press.

Walker, B. H. 2009. 'Foreword.' In *Brighter Prospects: Enhancing the Resilience of Australia*, ed. S. Cork. Canberra: Australia 21. pp. 2-3.

Walker, B. H., Abel, N., Anderies, J. M., and Ryan, P. 2009. 'Resilience, adaptability, and transformability in the Goulburn-Broken Catchment, Australia.' *Ecology and Society* 14 (1):12. Available from http://www.ecologyandsociety.org/vol14/iss1/art12/

Walker, H. D. 1985. 'An alternative approach to goal programming.' *Canadian Journal of Forest Research* 15:319-325.

Watson, A. 2007. A National Plan for Water Security: Pluses and minuses. Paper presented to the seminar series of the Victorian Branch of the Australian Agricultural and Resource Economics Society, 7 March, Melbourne.

Watson, W. D., Reynolds, R. G., Collins, D. J., and Hunter, R. D. 1983. Agricultural Water Demand and Issues. Water 2000: Consultants Report No. 5. Canberra: Department of Resources and Energy.

Weisbrod, B. A. 1964. 'Collective-consumption services of individual-consumption goods.' *Quarterly Journal of Economics* 78:471-477.

Williamson, O. E. 2000. 'The new institutional economics: Taking stock, looking ahead.' *Journal of Economic Literature* XXXVIII (September):595-613.

Wilson, M. A., and Hoehn, J. P. 2006. 'Valuing environmental goods and services using benefit transfer: The state-of-the art and science.' *Ecological Economics* 60:335-342.

Windle, J., and Rolfe, J. 2007. Developing a benefit transfer database for environmental values in Queensland. Paper presented to the 51st Annual Conference of the Australian Agricultural and Resource Economics Society, New Zealand.

World Commission on Dams. 2000. *Dams and Development: A New Framework for Decision-Making*. London: Earthscan.

Zografos, C., and Oglethorpe, D. 2004. 'Multi-criteria analysis in ecotourism: Using goal programming to explore sustainable solutions.' *Current Issues in Tourism* 7 (1):20-42.