
August 2010 Volume 91 Number 8 ISSN 0301-4797
ELSEVIER
Journal of
Environmental
Management

Journal of Environmental Management 91 (2010) 1613–1622

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Review

Embedded value systems in sustainability assessment tools and their implications

Alexandros Gasparatos [a,b,*]

[a] Institute of Advanced Studies (UNU-IAS), United Nations University, 1-1-1 Minato Mirai, Nishi-ku 220-8502, Yokohama, Japan
[b] School of Frontier Sciences, Department of International Studies, University of Tokyo, Japan

ARTICLE INFO

Article history:
Received 16 September 2009
Received in revised form
4 March 2010
Accepted 23 March 2010
Available online 22 April 2010

Keywords:
Sustainability evaluation
Participation
Human values
Value articulating institutions

ABSTRACT

This paper explores the implications that arise with the selection of specific sustainability evaluation tools. Sustainability evaluation tools are conceptualized in this paper as value articulating institutions and as such their choice is a far from a trivial matter. In fact their choice can entail various ethical and practical repercussions. However, in most cases the choice of the evaluation tool is made by the analyst (s) without taking into consideration the values of the affected stakeholders. By choosing the analytical tool the analyst essentially “subscribes to” and ultimately “enforces” a particular worldview as the legitimate yardstick to evaluate the sustainability of a particular project (or policy). Instead, this paper argues that the selection of evaluation tools should be consistent with the values of the affected stakeholders. With this in mind, different sustainability evaluation tools' assumptions are critically reviewed and a number of suggestions that could facilitate the choice of the most appropriate tool according to the context of the sustainability evaluation are provided. It is expected that conscious evaluation tool selection, following the suggestions made in this paper, will reduce the risk of providing distorted sustainability evaluations.

1. Introduction

A growing number of different evaluation[1] tools and frameworks have been developed in order to support conscious environmental decision making, e.g. (Deakin et al., 2007). These trends of evaluation tool/framework proliferation will most likely continue uninhibited as a result of three interconnected policy realities.

First of all it seems inconceivable, nowadays, to decide about major projects and policies without an analysis of the positive and negative impacts on the environment. Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA) have become legal requirements in several countries for projects/policies/etc that are expected to impact the environment significantly, e.g. through Directives 97/11/EC (EC, 1997) and 2001/41/EC (EC, 2001) in the European Union (EU). Of course the evaluation of projects and policies is not something new. Monetary tools, such as Cost Benefit Analysis (CBA), have been used for project and policy appraisal since the early 1970s, e.g. (HMSO, 1971) as cited by Glasson et al. (2005).

What is new though is the increasing demand for more and better quality evaluations.[2] A testament to this is the proliferation of national and regional professional evaluation societies (presently more than twenty), the increasing volume of relevant academic literature and most importantly the demand for accountability and freedom of information by national and international legislation. Currently, several organizations are providing guidelines and standards in order to assure the quality of an evaluation and thus its usefulness for decision making. Such examples include the Independent Evaluation Group of the World Bank (Mackay, 2007; World Bank, 2004), the Development Assistance Committee (DAC) of the OECD (e.g. OECD, 1998) and the European Commission [3] (EC, 2008a). Evaluation can also be seen as the process of providing information to stakeholders and as such is considered pivotal to emerging paradigms of

* Institute of Advanced Studies (UNU-IAS), United Nations University, 6F International Organisations Centre, Pacifico Yokohama, 1-1-1 Minato Mirai, Nishi-ku 220-8502, Yokohama, Japan. Tel.: +81 45 221 2343.
E-mail address: gasparatos@ias.unu.edu

[1] Despite certain subtle differences in their definitions, the terms evaluation and assessment are used interchangeably in this paper.

[2] Quality in evaluation is not universally defined as it may depend on the context of the evaluation, the methodology adopted and the client's needs. Nevertheless principles such as impartiality, independence, credibility and usefulness can be considered as some key aspects (OECD, 2007).

[3] Interestingly different departments of the European Commission have adopted different evaluation standards (Widmer, 2005).

0301-4797/$ – see front matter © 2010 Elsevier Ltd. All rights reserved.
doi:10.1016/j.jenvman.2010.03.014

governance such as adaptive governance (Dietz et al., 2003; Folke et al., 2005).

A second important factor has been the increasing prominence of sustainable development in the public dialogue. Sustainable development entered the public debate in the late 1980s after the Brundtland Commission report that defined it as "*development that meets the needs of the present without compromising the ability of future generations to meet their own needs*" (WCED, 1987). Since then it has become an important buzzword for policy makers, and hardly any policy discussion takes place at the highest levels without the term cropping up e.g. (DEFRA, 2005; EC, 2008b; UN, 1992). Sustainable development advocates make the case that a whole lot of different legitimate perspectives, sometimes conflicting (present vs. future generation interests, local vs. global interests, etc), are relevant and need to be considered during the decision making process. This "demand" to balance many different perspectives in a meaningful manner further complicates the decision making process.[4]

The third important factor is the increasing interest for public participation. Even though there has been a recurring demand for broader and more meaningful public participation, at least in the EU, which dates back to 1985 and the Directive on Environmental Impact Assessment (Directive 85/337/EEC) it was the prominence of sustainable development as a legitimate policy objective that further promoted it. Agenda 21 was instrumental in bringing more attention to the notion that environmental issues are better approached when all concerned parties are involved (UN, 1992). Since then public participation for environmental issues has become legally binding within the EU after the ratification of the Aarhus Convention on "*…[A]ccess to information, public participation in decision making and access to justice in environmental matters*" as the Directive 2003/35/EC (EC, 2003). As a result, meaningful public participation has become an overarching policy requisite and has found its way in numerous subsequent pieces of environmental legislation such as the Water Framework Directive (EC, 2000) and the Floods Directive (EC, 2007). Furthermore, participatory monitoring and evaluation have been suggested as useful procedures for the evaluation of development projects around the world (World Bank, 1996, 2002).

Considering the fact that future project/policy proposals will have to be decided within this nexus of evaluation, sustainable development, public participation and the increasing need for better quality evaluations, this paper aims to:

- provide a critical review of current evaluation tools and practices;
- elucidate the implications of certain methodological choices when evaluating the progress sustainability;
- make suggestions that can enhance the quality of sustainability evaluations.

In particular this paper focuses on the analytical part of a sustainability evaluation (Section 2) and offers suggestions that can reduce the risk of providing distorted sustainability evaluations. In the author's opinion one of the main reasons behind distorted sustainability evaluations is the tendency of analysts to employ evaluation tools in situations that these tools are not fit to be used. In order to appreciate the right context that each sustainability evaluation tool can operate it is of primary importance to understand these tools' main assumptions.

From this starting point this paper offers a critical review of the key assumptions within different sustainability evaluation tools and in particular of their embedded valuation systems and value judgments (Section 3). Section 4.1 illustrates how such evaluation tools can be conceptualized as value articulating institutions.[5]

More importantly the paper proceeds to discuss the implications and the risks that are associated with current evaluation tool selection practices. It is shown that in most cases the choice of the appropriate analytical tool is left to the analyst to decide – something that can have important repercussions to the quality and usefulness of the evaluation (Section 4.2). Finally the paper offers suggestions on how these findings can be operationalised and integrated within existing evaluation frameworks (Section 5).

The framing of the debate and the suggestions made herein can be of particular interest to evaluators that conduct environmental appraisals and sustainability assessments.

2. Defining key concepts

2.1. Tools vs. frameworks

It is important for the clarity of this paper to make a distinction between the notions of an evaluation tool and an evaluation framework. Frameworks are integrated and structured procedures, akin to protocols, which contain a number of prescribed stages that ought to be followed in order to meet a pre-determined objective. A key element of such assessment frameworks (e.g. the EIA or the SEA) is the comparison of the different project/policy alternatives based on their impacts to the environment. Even though such assessment frameworks are legal requirements they do not specify the different analytical tools that must be used for the analysis of the different alternatives.

For the purpose of this paper evaluation tools are defined as the various analytical techniques that can be used to conduct analyses/comparisons within frameworks such as the EIA or the SEA.

Such evaluation tools attempt to understand a system and offer information in a format that can assist the decision making process (Gasparatos et al., 2009). This is usually done by quantifying certain aspects that are deemed relevant (e.g. monetary costs/benefits, resource consumption, environmental impact etc) and in most cases aggregating these aspects. The most widely utilised evaluation tools include economic tools (e.g. CBA, Whole Life Costing), biophysical models (e.g. Material Flow Analysis, Ecological Footprint, Energy Accounting), indicator lists/composite indices and Multi-Criteria Analysis (MCA). Although these tools are routinely referred to as environmental appraisal tools (e.g. Vatn, 2009), all of them can capture different aspects of sustainability (e.g. Gasparatos et al., 2008, 2009; Ness et al., 2007; Stagl, 2007). So for the remainder of the paper these tools will be referred to as sustainability evaluation tools.[6]

The evaluation tools mentioned above fall within two broad categories; reductionist and non-reductionist. Economic tools,

[4] The interested reader is referred to Gibson et al. (2005), Pearce (1993) and Pope et al. (2004) among numerous others for definitions of sustainable development and key methodological aspects and challenges of sustainability assessment.

[5] According to Vatn (2005a: 211) a value articulating institution is "*…a constructed set of rules or typifications. It defines who shall participate and on the basis of which capacity — i.e., in which role*" and "*…what is considered relevant data and how data is to be handled*" Refer to Section 4.1 for an extended discussion of the concept and its relevance to this paper.

[6] It should be noted that these tools can be used to evaluate the performance of projects and policies. However biophysical and indicator tools have mainly been used to assess the performance of projects and as a result the paper will focus on the sustainability evaluation of projects. Nevertheless most of this paper's findings and conclusions can also be extended and for the evaluation of policies.

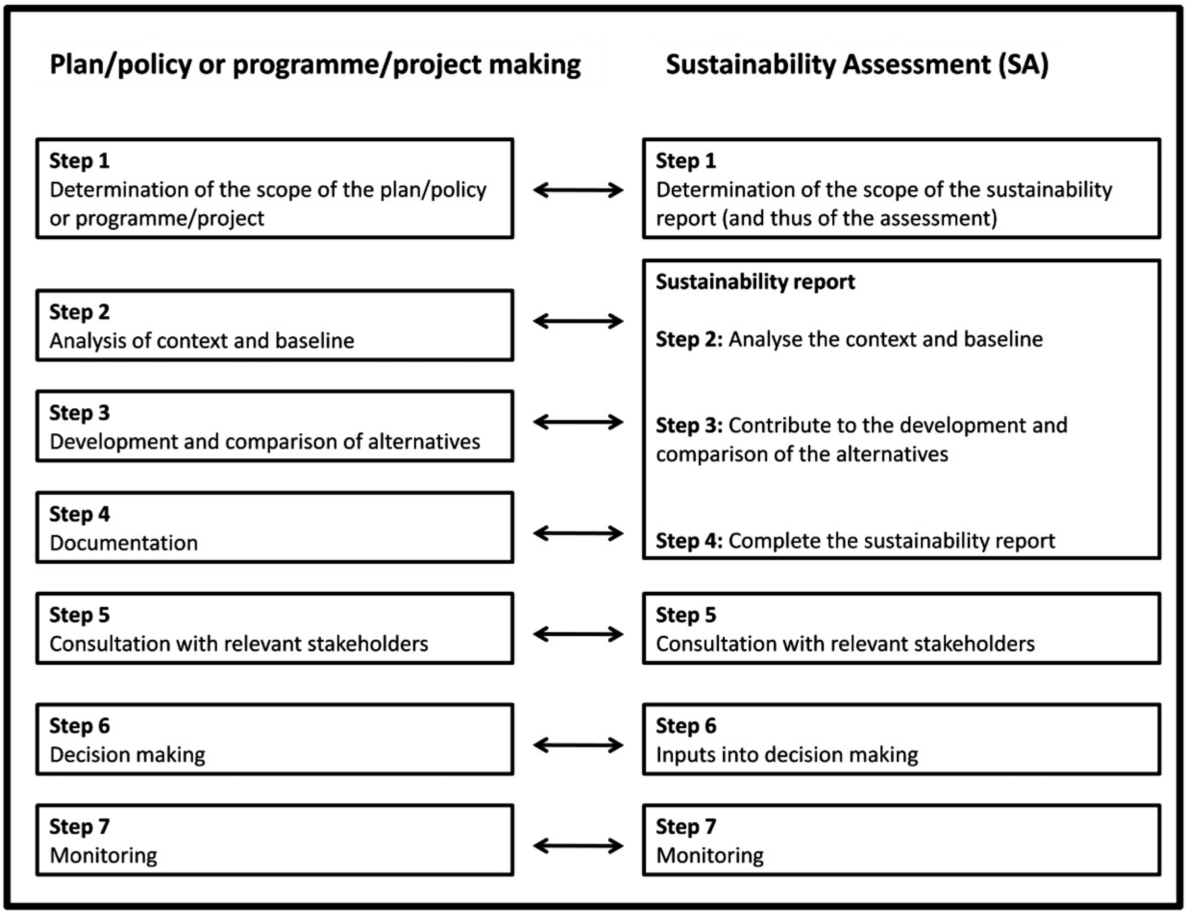

Fig. 1. Links between plan/policy and programme/project making tasks and Sustainability Assessment (SA) elements, adapted from UNECE (2007: p. 28). Note: Public participation and consultation with authorities are not included in full in this simplified diagram (refer to the Supplementary material for a more detailed description).

biophysical models and composite indices fall mainly within the former category while MCA falls within the latter.

To appreciate these subtle differences between evaluation tools and frameworks the SEA will be used as an example.[7] According to UNECE (2007) the purpose of a SEA "*…is to ensure that environmental considerations inform and are integrated into strategic decision making in support of environmentally sound and sustainable development*".[8] In order to achieve this, a number of methodological steps have been prescribed, i.e. scoping, analysis of baseline, development/comparison of alternatives etc (refer to Fig. 1).

An integral part of the whole process is the development and comparison of different alternatives as well as the assessment of their performance (Step 3, Fig. 1). Nevertheless no analytical technique is being singled out as the most appropriate for this task. In fact, it is acknowledged that there is no single "best" analytical technique, but its choice is important for the quality of the information provided to the decision makers (UNECE, 2007). In a way the choice of the appropriate analytical tool is left to the analyst, something that can have important repercussions to the overall procedure as will be discussed in Section 4.2.

This paper focuses on certain decisions that are being made in Step 3 of the generic evaluation framework highlighted in Fig. 1.

2.2. Valuation systems vs. human values

A second key distinction is between the concepts of value (valuation systems) embedded in evaluation tools and the human values which are inherent human characteristics.

Human values refer to "*…a desirable trans situational goal varying in importance, which serves as a guiding principle in the life of a person or other social entity*" (Schwartz, 1992). Karp (1996) argues that human values are relatively few in number, are relatively stable, are guides for behavior and they can have a measurable influence on behavioral choice.[9]

Milfont et al. (2006) mentions that altruistic and individualistic value orientations imply that people will judge environmental issues on the basis of costs/benefits (in the broad sense) to human groups on the former case or an individual in the latter case. On the contrary, people with biospheric value orientations in most cases judge and act primarily thinking of the impact to the environment (ibid). Similarly, Stern et al. (1995) suggested that pro-environmental action depends on both beliefs and values and identified three relevant types: the social-altruistic value orientation (concern

[7] The choice of a Sustainability Assessment framework would have been more appropriate for the purpose of this paper. However to the author's best knowledge no such framework has become a standard or a legal requirement nationally or internationally.

[8] It is interesting to note how the concepts of evaluation, participation and sustainable development that were introduced in the previous section are linked with respect to environmental decision making.

[9] A review of different definitions of value, typologies, origins and their impact on human behavior are provided by Hitlin and Piliavin (2004).

for other human beings) the biospheric orientation (concern for nonhuman species) and self interest (concern for the self).

In the same sense that human values are inherent characteristics of humans, valuation systems are inherent characteristics of certain evaluation tools. Valuation systems, or concepts of value as are sometimes encountered in the academic literature (e.g. Patterson, 1998) are essentially some of the deepest assumptions of evaluation tools and in a sense indicate what these tools "consider" important and how it is to be measured. An evaluation tool's concept of value essentially mirrors the worldview that the tool "embraces" when measuring the sustainability performance of projects. Section 3.1 provides an overview of two main valuation systems found within evaluation tools, i.e. subjective preference theory of value and cost of production theory of value.

3. Evaluation paradigms and embedded value systems

What is really important within the argumentation of this paper is to appreciate that each tool employs a different worldview when measuring the performance of a project. In order to understand how the choice of a tool will affect the outcome of the evaluation, the basic assumptions behind each tool are reviewed.

3.1. Reductionist tools

According to Munda (2006), reductionist tools make use of a single measurable indicator (e.g. economic costs/benefits), a single dimension (i.e. one of the economic, environmental or social dimensions of sustainability), a single scale of analysis, a single objective (e.g. maximization of economic efficiency) and a single time horizon. The advantage of such tools lies in the fact that they can measure the performance of projects by reducing and integrating their diverse aspects to a small set of numbers. This approach is to a certain degree compatible with the desire to "keep it simple" usually expressed by stakeholders and policy makers.

In order to do so, economic tools and biophysical models adopt common denominators (currencies). However, economic and biophysical tools employ different concepts of value (Ecological Economics, 2002; Gasparatos et al., 2009; Patterson, 1998) so their evaluations employ radically different perspectives.

Monetary tools tend to capture a person's Willingness To Pay (WTP) for the consumption of a commodity or a person's Willingness To Accept (WTA) compensation to forfeit the consumption of a commodity. WTP and WTA are in turn considered as proxies for the measurement of a person's utility (Farber et al., 2002; Layard, 2005). Essentially economic tools based on the neo-classical paradigm account for the difference that consumption or non-consumption makes to the satisfaction of human preferences and in a way maximizing consumption is equated with maximizing happiness is (Ayres et al., 1998; Layard, 2005). Or as Spash (2007a: p. 691) puts it "…*in a standard market setting individuals engage in selling their labour and buying consumer items and their own limit on obtaining happiness is their ability to pay*". As a result "…*the basic notion of value that guides economic thought is inherently anthropocentric*" (Farber et al., 2002: p. 379). Patterson (1998) refers to the theory of value adopted by economic tools as subjective preference theory of value. Additionally, there are certain criticisms within CBA which are concerned with the compensability and subsequent substitutability of the monetised quantities that represent the different socioeconomic and environmental issues included in the evaluation. This implies the existence of trade-offs between different sustainability issues (Munda, 1996) which are at the core of the debate of strong vs. weak sustainability (Neumayer, 2004), refer to Section 3.2 for an example.

Biophysical models, on the other hand, account for the amount of resources in the broadest sense that have been invested for the production of a good/service.[10] They assume that the single most important yardstick when evaluating projects is the amount of natural resources appropriated, as a proxy to environmental impact. According to biophysical tools' assumptions the fewer resources appropriated, the more agreeable the project is.[11] This is similar to the cost of production theory of value (Patterson, 1998). The fact that biophysical models adopt a different concept of value (valuation system) implies that they employ a different valuation perspective (eco-centric) that is different to that of standard economic analysis (anthropocentric perspective). This has been discussed extensively in the academic literature and results in the biophysical tools' general tendency to neglect certain human preferences (Cleveland et al., 2000; Winkler, 2006).

According to Neumayer (2004) biophysical models follow the spirit of strong sustainability without being direct measures of it perhaps because biophysical tool actually measure only one type of capital, natural capital. Even when these tools measure issues that could be classified as economic or social, thereby falling under the category of social/economic capital, they measure them through the appropriation of natural capital. Typically there is a substitution between different types of natural capital within biophysical tools given their highly aggregated nature. Nevertheless the trade-offs essentially happen between the same form of capital (natural capital) and not between different forms of capital. Weak sustainability by definition refers to the substitution between different forms of capital (Neumayer, 2004). Hence, biophysical tools cannot be considered weak sustainability tools while at the same time they are not purely strong sustainability tools.

A composite indicator/index can be simply defined as an aggregation of different indicators under a well developed and predetermined methodology. In contrast to monetary tools and biophysical models, composite indices do not make use of a common denominator and as a result they do not have an explicit valuation system. In fact any concept of value is lost during the normalization stage [refer to OECD, (2008) for an overview of the architecture of composite indicators]. However some of the methodological choices during their construction are value judgments in their own right. For example, choices regarding the indicators (not the issues represented) that will constitute the composite index are usually made by the analyst(s). Nevertheless, indicator choice can have a significant impact on the outcome of the evaluation. Lee (2006) points out that if the aim of the analyst is to address intergenerational equity then stock indicators would be more appropriate. Conversely in situations that the analyst seeks to

[10] Biophysical sustainability assessment tools such as emergy synthesis, exergy analysis and the ecological footprint have been used in cases as diverse as residential heating systems (Zmeureanu and Wu, 2007. R. Zmeureanu and X.Y. Wu, Energy and exergy performance of residential heating systems with separate mechanical ventilation, *Energy* 32 (2007), pp. 187–195. Article | PDF (177K) | View Record in Scopus | Cited By in Scopus (9)), waste management (Sciubba, 2003; Yang et al., 2003), wetland management (Ton et al., 1998), cropping systems (Lefroy and Rydberg, 2003) and fuels (Holden and Hoyer, 2005).

[11] Generally speaking biophysical models attempt to capture the environmental support required by a project. Even though it is not overtly evident, central to these tools' philosophy is the "vision" of minimizing human impact on the biosphere (Gasparatos et al., 2008). According to this "vision" when project alternatives are assessed then the alternative with the lowest Ecological Footprint or non-renewable resource consumption (for material flow analysis, exergy analysis, etc) would be the most acceptable option as this would imply lower resource utilization and in some cases a lower environmental impact. This choice is "made" regardless of human preferences and might result in favor of a project with higher cost-to-benefit ratio, something unacceptable under standard economic analysis.

capture intra-generational equity then a choice of distributional indicators would seem more justified. Another important issue is the weighting of the indicators. Weights are value judgments, so "…*greater weight should be given to components which are considered to be more significant in the context of the particular composite indicator*" (OECD, 2003: p. 12). Even though it is becoming standard practice to consult stakeholders during the assignment of the weights there are still cases where the weights are designated by the analysts themselves. This is not necessarily bad but it may compromise effective public participation and have negative implications regarding the acceptability of the evaluation results (refer to Section 4.2).

Perhaps the most important decision during the construction of the composite index that ultimately affects the outcome of the whole evaluation is the final aggregation of individual indicators. Given that the aggregation of different indicators implies trade-offs between the issues represented, then it can be inferred that composite indicators are measures of weak sustainability like CBA.[12] The most popular aggregation techniques are the linear aggregation that allows unlimited trade-offs between issues and the geometric aggregation that allows trade-offs, albeit to a lesser degree. However the choice of aggregation and the acceptability/extent of trade-offs it implies is a choice made by the analyst and is a value judgment in its own right that does not necessarily reflect the views of stakeholders. For example, in certain cases stakeholders might decide to intensify the exploitation or even alter an ecosystem in order to meet more urgent needs such as to alleviate poverty or offer protection from natural disasters such as flooding (trade-offs are acceptable). Alternatively there can be a conscious choice to sacrifice certain economic activities, and thus some of the short term economic welfare of the community in order for example to maintain the ecosystem services provided by a fragile ecosystem (trade-offs are unacceptable).

These criticisms and inherent value judgements within reductionist tools are only some of the most important. Gasparatos et al. (2008, 2009) discuss in depth how the assumptions and methodologies of these tools can sometimes be inconsistent with certain aspects of sustainable development and what different aspects of sustainable development each tool illuminates.

3.2. Non-reductionist tools

MCA is a family of indicator-based techniques whose development follows a procedure similar to that of composite indices.[13] As a result several of the methodological choices including the choice of the indicator and of the weights are "subjective" (e.g. data selection, criteria definition, aggregation and weighting (Messner et al., 2006)) which are influenced to a great degree from the analyst as discussed earlier. van den Hove (2006: p. 15), for example, writes that in certain MCA tools there have been assigned "*equal weights to the dimensions used in the analysis (e.g. economic, social, environmental dimensions) notably because it "may reduce the probability of social conflicts inside the scientific team"*". In a similar manner Munda (2006: p. 89) acknowledges that in his study "*all the indicators are considered to have the same importance, i.e., no weighting coefficient is used*".

The main important difference is that in MCAs, contrary to composite indicators, an aggregation of the individual indicators does not take place. This is in order to prevent trade-offs between them and in effect between the different sustainability issues represented. This is closer to the concept of strong sustainability as discussed by Gasparatos et al. (2008) and it is, once more, a value judgment made by the analyst that might not necessarily reflect stakeholders' considerations.[14]

4. Understanding the problem

4.1. Conceptualising evaluation tools as value articulating institutions

As it becomes obvious from the previous sections, different sustainability evaluation tools make different assumptions on what is important to measure and how to measure it. These assumptions are, in their core, constructed sets of rules/typifications which are at the same time embedded value judgments. As a result the outcome of such sustainability evaluations is far from value-free and neutral. It is thus justified to claim that when a certain tool is chosen as a yardstick to measure the performance of a project, at the same time a value-laden evaluation perspective is inevitably also chosen.

In this sense these sustainability evaluation tools are meeting the first component of the definition of value articulating institutions (Jacobs, 1997; Vatn, 2005a,b, 2009), refer to Footnote 5 for this definition. Value articulating institutions also define (a) "*who and in which capacity, i.e. in which role*", should be considered during the decision making process and (b) "*what is considered relevant data and how data is to be handled*" (Vatn, 2005a: p. 211).

The latter point is the most relevant for this paper and was discussed in the previous section. However, the former point is also of some interest. Neo-classical economic valuation tools such as the Contingent Valuation Method (CVM)[15] view human as individual consumers that try to maximize their utility (Heinzerling and Ackerman, 2002) and that this "*net utility from the consequences of an action determines whether that action is right or wrong*" (Spash et al., 2009). Deliberative Monetary Valuation (DMV)[16] approaches, on the other hand, view humans as citizens or parts of broader social groups which unequivocally affects their attitude including the valuation of environmental, e.g. refer to (Howarth and Wilson, 2006; Niemeyer and Spash, 2001; Sagoff, 1998; Spash, 2007a; Wilson and Howarth, 2002) for theoretical discussions and (Spash, 2007b, 2008) for a review of empirical studies.

On the other hand, when biophysical models are utilized then the role of the human seems to become altogether obsolete as these tools seem to neglect human preferences (Cleveland et al., 2000; Winkler, 2006). Although biophysical tools seem to focus on the "interest" of nature at the same time they seem to indirectly cater for future generations. As a rule of thumb projects with the lowest

[12] Aggregation implies compensability between indicators and which in turn implies substitutability between the different sustainability issues that the indicators represent. The moment that aggregation takes place then the weights lose their meaning of value judgment and obtain the meaning trade-off ration (Munda and Nardo, 2005). This phenomenon is not a result of how weights are assigned but of the aggregation rule used, most commonly linear aggregation.

[13] MCA has been used for water management (De Marchi et al., 2000; Kallis et al., 2006; Salgado et al., 2009) and renewable energy assessment (Gamboa and Munda, 2007; Madlener and Stagl, 2005) among others.

[14] Vatn (2005a: 210–211) argues that given its lack of common currency, MCA, also implies that respondents are accepting values to be expressed in different, even incommensurable dimensions.

[15] Sustainability issues that have been quantified with CVM include urban green spaces (Tyrvainen, 2001; Jim and Chen, 2006), water quality (Atkins and Burdon, 2005; Bateman and Turner, 1993), road noise (Fosgerau and Bjorner, 2006), biodiversity (Christie et al., 2005), health (Habbani et al., 2006; Johannesson et al., 1996; Mataria et al., 2006), culture (Dutta et al., 2007; Thompson et al., 2002) and education (Stair et al., 2005).

[16] DMV attempts to combine large-scale stated-preference methods (e.g. CVM) with small scale group deliberation (Spash, 2008). According DMV theorists, DMV is "…*a social process of valuation because it engages individuals as representatives of social groups*" (Spash, 2008: p. 471).

biophysical value (particularly for non-renewable resources) are more agreeable as mentioned in Footnote 11. This is an indication of resource appropriation which seems to imply that by consuming fewer resources now, more resources are left in place for future generations to realize their potential.

4.2. Implications of current tool selection practices

The choice of the evaluation tool (and of certain methodological steps within tools), and hence the worldview through which the evaluation is performed is in most cases made by the analyst(s) and is rarely, if ever, influenced by the stakeholders' views, needs and values. But what happens in those cases that the value judgments embedded in an evaluation tool are not aligned with the values of the affected stakeholders? Two interesting consequences arise, an ethical and a practical one.

The ethical implication lies in the fact that by choosing a certain tool to evaluate a project, the analyst "subscribes to" and in effect "enforces" a specific worldview as the correct or most appropriate yardstick to measure the performance of a project from which he/she is not, most likely, going to be directly affected. However, the role of the analyst as an all knowing expert and the choice of one legitimate perspective above all others have drawn significant criticism (Funtowicz and Ravetz, 1993). This deliberate (or accidental) bypassing of the stakeholders' views and values can, in a sense, be contrary to the whole idea behind meaningful public participation. A by-product of this is that the analyst by choosing the tool "assumes" willingly or unwillingly the role of the stakeholder in his/her own right.

It should be clarified here that the analyst might not necessarily agree, share, or even be aware of the value judgments that a specific tool embraces. However, this worldview (what is considered relevant data, who and in what role is considered in the analysis, what legitimate perspectives of sustainability are captured etc) is embedded in the assumptions of the tool and as a result is an attribute of the tool that exists regardless of the analyst. It is fair to say that the moment the tool is used these attributes enter the sustainability evaluation and unequivocally affect it.[17]

The practical implication is more relevant to the fact that the performance of the project is not measured in a way that mirrors the needs and expectations of end users. For example, a housing project might be built following the most stringent environmental standards but what happens if for any number of reasons people are not too keen to live in? Despite its good environmental performance if this building is abandoned before the end of its projected life cycle it can still be perceived as a waste of natural resources as the investment of natural resources has not fulfilled its initial objective. Vatn (2005a) makes a similar comment by making the case that when using different value articulating institutions different evaluation results and preferred solutions arise. He goes even further to suggest that "…*the people involved in the process will most probably have ideas about which frames they find most relevant. One should even expect that they would react negatively to frames that are not found relevant or acceptable – e.g. using individual-oriented value articulated institutions for something considered collective and vice versa*" (Vatn, 2005a: p. 211). In a similar fashion Spash (2000: p. 205) suggests that "…*individuals may reject the institution that imposes such a condition on them*".[18] Niemeyer and Spash (2001: p. 567) have argued that the most appropriate approach to policy making involves a difficult act of balancing performance in theory and practice and that "…*if the theoretical basis is ontologically flawed – too abstract, ignoring important components, or factually incorrect – the method will fail to produce theoretically desired outcomes*".

This practical implication can affect the long-term sustainability of projects and can have repercussions on both stakeholders, at various scales, as well as the analyst(s). On the one hand the users and other directly affected stakeholders might end up with projects that could be inconsistent with their needs and values. It is doubtful whether such projects will realize their full potential and be sustainable in the long run. In a similar manner, decision makers such as governments and corporations might face considerable scrutiny over their decision to go ahead with such projects something that might materialize as political cost in the former case and economic loss in the latter. On the other hand evaluators that might come from academia, government or consultancy also have a great stake not to offer distorted evaluations as they are putting their reputation on the line. This could manifest for example in the decrease in future earnings of a professional evaluator after providing poor evaluation results.

5. Reducing the risk of providing distorted sustainability evaluations

The previous section has suggested that understanding the values, attitudes and needs of the affected stakeholder can provide valuable insights during a sustainability evaluation. From this starting point it is suggested that stakeholder values can be a good guiding principle for choosing the most appropriate evaluation tool.

Of course it seems utopian to demand stakeholders to directly choose the analytical tool which reflects their own needs and values in the most effective manner. Although stakeholders, even with minimal education, have shown to grasp advanced technical information and models when confronted with them (e.g. Becu et al., 2008), and despite the notion of extended peer community catching on (Funtowicz and Ravetz, 1993), the fact remains that the complicated nature of such tools will most likely render this direct attempt to consider stakeholder values not feasible. Knowledge of the intricacies and the mechanisms within these tools, coupled with the fact that most tools trace their roots in different branches of knowledge, requires specialist training and expertise.

Additionally, even the choice of certain methodological steps within the evaluation tools themselves, e.g. the choice of the most appropriate non-market valuation technique for CBAs or the most appropriate indicators selection/aggregation/etc rules in indicator-based tools is challenging and requires significant expertise in order to be theoretically sound and fit for the purpose of the evaluation.

5.1. Considering human values when choosing an evaluation tool

From the previous sections it seems justified to claim at this point that:

(a) the different valuation systems embedded in tools can reflect certain value orientations found in humans;
(b) the valuation system of the selected tool should reflect the value orientation of the affected stakeholders.

Ojea and Loureiro (2007) have shown that individuals with egoistic and altruistic value orientations had a higher probability of paying for a wildlife restoration initiative in a CVM study. This conclusion agrees with previous findings by Stern et al. (1993, 1995) which show the reduced importance of biospheric value orientation, on the one hand, and the increased importance of anthropocentric

[17] For example Vatn (2005a: p. 210) states that the type of institutions that are invoked in the process of expressing value, influences both the values that come forward and the conclusions that can be drawn on the basis of them.

[18] An example is the refusal of some respondents to provide WTP values in CVM surveys for the protection of wildlife (or WTA for the destruction of wildlife) despite their beliefs that wildlife ought to be preserved (e.g. Lockwood, 1998; Spash, 2000; Spash and Hanley, 1995).

Table 1
Summary of evaluation tool features.

Tool	Concept of value (valuation system) Sections 3.1–3.2	Valuation perspective Section 3.1–3.2	Role of participant Section 4.1	Relevant stakeholder value orientation Section 5.1
Biophysical	Cost of production [a]	Eco-centric	Participant becomes irrelevant	Biocentric
Traditional monetary valuation (e.g. CVM)	Subjective preference [a]	Anthropocentric	Individual consumer	Egoistic
Deliberative monetary valuation (DMV)	Inconclusive evidence[b]	Anthropocentric	Citizen	Altruistic
Composite indicators	Lost during normalisation and aggregation	Lost during normalisation and aggregation	Lost during normalisation and aggregation	Lost during normalisation and aggregation
Multi-Criteria Analysis (MCA)	Depends on methodological choices [c]	Depends on methodological choices [c]	Depends on methodological choices [c]	Depends on methodological choices [c]

[a] Following the distinction made by (Patterson, 1998).
[b] Refer to Vatn (2005b) and Spash (2008). Howarth and Wilson (2006) have suggested that in DMV additional concerns to economic efficiency (e.g. fairness of distribution) are expected to be articulated.
[c] Particularly on indicator selection and weighing.

orientation (egoistic/altruistic), on the other hand, when determining economic behavior. Such findings have been corroborated to a degree by Spash (2006). In his CVM study it was found that the "…*egoistic-altruistic attitudes were more likely to be associated with positive bids than protests*", that the "…*egoistic-altruistic attitudes scale proved to be highly significant and explained more of the variance in WTP than any other variable*", that the "…*social biospheric scale was weaker and proved to explain little of the variance beyond that covered by egoistic-altruistic attitudes and rights based beliefs*" and that the "…*egoistic and selfish altruistic motives are important determinants of WTP*" (Spash, 2006: p. 620).

The above finding seem to imply that economic tools might be appropriate tools for sustainability evaluations where stakeholders hold anthropocentric value orientations (egoistic and altruistic) while they might be inappropriate tools when stakeholders hold biospheric value orientations. On the other hand a biospheric value orientation (refer to Section 2.2) means that an evaluation using biophysical tools would be more consistent with the perception of the stakeholders.

Egoistic or altruistic value orientations can further affect methodological choices within monetary analysis as well. In the former case traditional valuation techniques such as CVM might be more appropriate, while DMV seems more appropriate in the latter case. This of course stems from the different role that the participants assume in neo-classical economic valuation exercises (i.e. as individual consumers in CVM surveys) and in DMV (i.e. as citizens) – refer to Section 4.1. Furthermore, it has been shown that consumer values are better addressed by CVM while such a tool was deemed as inappropriate to address the citizen values (e.g. Spash, 2000). Additionally WTP estimates are related with egoistic value orientations, and not with altruistic values (Stern and Dietz, 1994). Finally, Spash (2008) makes the case that most DMV theorists advocate group arbitrated values which further implies that DMV might be the appropriate tool when stakeholders exhibit altruistic value orientations.

In the same manner, understanding the value orientation of the affected stakeholders can suggest which indicators would be more appropriate in composite indices and MCAs as well as the most appropriate way to design the composite index and assign weights/normalise, etc.

Table 1 summarises the main features of the different evaluation tools as discussed in (Sections 3.1, 3.2, 4.1 and 5.1).

5.2. Other considerations

Understanding certain stakeholder attitudes can also offer helpful insights for the selection of the most appropriate evaluation tool and as such to help reduce the risk of providing distorted evaluations. Two key attitudes are

- the acceptability and the extent of trade-offs (weak vs. strong sustainability perspective). Acceptability of trade-offs would mean that it is theoretically sound to use CBA which allows unlimited substitution between the monetized values that are fed in it. Furthermore, it could inform the type of aggregator in indicator-based tools with linear aggregation, for example, being relevant if unlimited trade-offs are acceptable, geometric aggregation when limited trade-offs are acceptable or no aggregation at all (MCA) when trade-offs are deemed unacceptable.
- the extent to which there is a strong feeling for preserving the rights of future generations. Knowledge on such issues should feed into the choice of right indicators as discussed in Section 3. Additionally, Gasparatos et al. (2008) discuss the different aspects of inter- and intra-generational equity that each tool captures. A better understanding of the aspects of equity which are relevant to the stakeholders can also be important for the choice of the appropriate tool.

5.3. Capturing values and attitudes

It has been made clear by now that it seems methodologically pressing to capture the values of the affected stakeholder in order to inform the decision of the most appropriate evaluation tool(s). There is a rich literature on how (e.g. through questionnaires, interviews, psychological research, etc)[19] to capture these values but the choice of the most appropriate approach should lie on the analyst(s). More importantly this choice should be made subject to the context of the case and be informed by other constraints such as the available time, expertise and budget.

Another important question that arises is at which point of the evaluation procedure (framework) should, the stakeholder values and attitudes towards the proposed alternatives (i.e. projects), be captured. This would depend on the format of the participatory process and must be judged by the evaluation team considering the context of the case, e.g. the degree of participation [consultation, partnership, delegated power etc refer to Arnstein, (1967)], the participatory method (e.g. focus group,

[19] Approaches such as the New Environmental Paradigm (NEP) e.g. (Dunlap and van Liere, 1978; Dunlap et al., 2000), the value system devised by Schwartz (Karp, 1996; Stern and Dietz, 1994) or the Value-Belief-Norm theory (BNF) (Stern et al., 1999; Stern, 2000) can be powerful tools when determining whether stakeholders hold biocentric, egoistic or altruistic values.

citizen juries, etc), whether the stakeholders are the decision makers themselves or not, etc.

However, it seems reasonable to suggest that the attitudes and values of the stakeholders should be captured at Step 2 or 3 of a sustainability assessment (refer to Fig. 1 and the Supplementary material). Stakeholder values can be captured during Step 2 if they are considered as context of the plan/policy or programme/ project. In that case they can be used to influence both the choice of the most appropriate evaluation tool as well as the development of the alternatives. Alternatively stakeholder values can be captured during Step 3. In this case they can be used only to influence the choice of the most appropriate evaluation tool.

Even though the stakeholders' overall value orientation will not change dramatically, at least in the short term, stakeholders are known in certain cases, to change their minds regarding the relative merits and the appeal of specific projects after information has been presented. Such an example has been the case of citizen's juries where the acceptance of different alternatives changed after information on their relative merits and drawbacks was provided (Niemeyer, 2005). In those cases perhaps a value capturing mechanism that explicitly links the values and attitudes of the stakeholder to the specific project, similar to the one used by Spash et al. (2009), might be more appropriate.

5.4. Evaluation team

Implementing the proposal discussed in the previous sections means that the evaluators should actively attempt to capture and employ the values of the stakeholders as guiding principles for the selection of the most appropriate evaluation tool. Multidisciplinary teams are essential in order to conduct such types of evaluations given that insights from different academic disciplines become relevant. Even though there seems to be a trend towards multidisciplinary evaluation groups what seems to be lacking is the inclusion of voices from branches of knowledge perceived as “soft”. Considering the importance of capturing the stakeholder values it might be valuable for such teams to include persons knowledgeable in psychology and anthropology that are not usually represented in professional evaluation teams.

5.5. Caveats and future research

Section 3 highlighted the different concept of value employed by monetary and biophysical evaluation tools (i.e. cost of production value and subjective preference value). On the other hand, the value orientations that stakeholders might have (i.e. altruistic, egoistic, biospheric) and the behaviours that might arise have been discussed in Section 2.2.

The combination of these two strands of academic literature (Table 1 summarises the main points) is made through their comparative review. This culminates with the suggestion of employing stakeholder values as a guide for selecting the most appropriate analytical tool during sustainability evaluations. As a result the proposal that emerges from this literature review should be currently viewed as a theoretical inquiry and a basis for future research. Gaps, especially on empirical cases, constitute the three major concerns regarding the applicability and expected benefits of this proposal and should be the focus of future research.

Caveat #1: it has been shown that people can often act in a specific manner according to their value system (e.g. Hitlin and Piliavin, 2004). Thus, it is argued that considering stakeholder values during the evaluation following the distinct typologies adopted in this paper may reduce the risk of providing a distorted evaluation through the random choice of an analytical tool. However, to the author's best knowledge there have not been any attempts in the academic literature to base the choice of the analytical tool on the value systems of the stakeholders.[20] As a result the assertion that the selection of the evaluation tools should be influenced by the values of the stakeholders cannot be empirically proven. Additionally there is some evidence which supports the existence of mixed motives and plural values (e.g. Spash, 2006), without however contradicting the basic premises of Stern et al. (1995) as discussed in Section 5.1.

Caveat #2: the possibility of all stakeholders having the same values is almost negligible, especially when increasing the spatial scale. In this case more than one tools or a well-balanced indicator approach should be employed in order to be more representative of this value multiplicity. Nevertheless the author is not aware of any significant research that could facilitate the linkage of biophysical and monetary tools in a meaningful manner.

Caveat #3: most of the studies regarding the links between values and observed behavior (e.g. pro-environmental behavior) have been conducted in developed countries, e.g. (Karp, 1996). The question then becomes whether such value orientations, e.g. biospheric, will give rise to the same kinds of behavior across different cultures. Empirical studies (e.g. Chung and Poon, 2001; Leung and Jenni, 2002; Milfont et al., 2006 among others) have shown that this can be the case between different groups within the same society (urban vs. rural populations in China, Anglo vs. Chinese Australians and European vs. Asian New Zealanders respectively). However, studies in different country contexts had mixed results (e.g. Bostrom et al., 2006; Gooch, 1995; Rauwald and Moore, 2002; Schultz and Zelezny, 1998, 1999). In a similar manner most of these studies are conducted considering only specific social groups, usually students, and as such their universal applicability is further questioned. Nevertheless, Janssen and Ostrom (2006) mention a number of cases where social science research findings, not exclusively related to environmental values though, have been replicated across different cultures and social groups.

6. Conclusions

The monetary, biophysical and indicator-based sustainability evaluation tools reviewed in this paper can be perceived as value articulating institutions. Such tools exhibit different embedded value judgments that in a sense articulate what is considered relevant data/how data is to be handled/who and in which capacity (i.e. role) should be considered during the decision making process/etc. These embedded value judgments can affect the outcome of the evaluation.

In most cases the choice of an evaluation tool is made by the analyst(s) without taking into consideration the needs, wants and values of the stakeholders. It is argued throughout this paper that such a choice is not trivial not the least because it carries ethical and practical implications. By choosing the tools the analyst(s) “subscribe(s) to” and ultimately “enforce(s)” a particular worldview as the legitimate yardstick to measure the performance of a project which might be incompatible with the worldview of the affected stakeholders. As a result the performance of a project is not necessarily measured in a way that mirrors stakeholders' values and possibly their needs and expectations. This practical implication can be detrimental to the long-term sustainability of projects and can have repercussions both to stakeholders and analyst(s).

It is theorized that the values of the affected stakeholders should guide the selection of the appropriate sustainability evaluation tool and it is suggested that certain stakeholder value orientations are

[20] Stakeholder values have been considered in some cases for the selection of the most appropriate indicators during regional sustainability assessments (e.g. Bell and Morse, 2004; Wallis, 2006; Wallis et al., 2007).

more compatible with the worldview of certain evaluation tools. For example, when stakeholders exhibit biospheric value orientation then evaluations employing biophysical tools seem more appropriate. On the other hand when egoistic and altruistic value orientations are prevalent then monetary tools using neo-classical valuation tools (e.g. CVMs) in the former case and DMV in the latter would be more appropriate.

Acknowledgments

The author would like to acknowledge the support of the Canon Foundation in Europe and the Japanese Society for the Promotion of Science (JSPS). A previous draft of this paper has been presented at EASY-ECO 2009. Helpful comments have been provided by Malcolm Horner, Klaus Hubacek, Katarina Soltesova and two anonymous reviewers. Much of this work has been inspired through discussions with lecturers and participants of two summer schools organized by the European Society of Ecological Economics (THEMES 2008 and 2009) and particularly with Arild Vatn and Sigrid Stagl.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.jenvman.2010.03.014.

References

Arnstein, S., 1967. A ladder of citizen participation. J.A.I. Planners 35, 216–224.
Atkins, J.P., Burdon, D., 2005. An initial economic evaluation of water quality improvements in the Randers Fjord, Denmark. Mar. Pollut. Bull. 53, 195–204.
Ayres, R.U., van den Bergh, J.C.J.M., Gowdy, J.M., 1998. Viewpoint: Weak Versus Strong Sustainability. Discussion Paper #98-103/3. Tinbergen Institute, Rotterdam. Available at: http://www.tinbergen.nl/discussionpapers/98103.pdf.
Bateman, I.J., Turner, R.K., 1993. Valuation of environment, methods and techniques: the contingent valuation method. In: Turner, K. (Ed.), Sustainable Environmental Economics and Management: Principles and Practice. Belhaven Press, London, pp. 120–191.
Becu, N., Neef, A., Schreinemachers, P., Sangkapitux, C., 2008. Participatory computer simulation to support collective decision-making: potential and limits of stakeholder involvement. Land Use Pol. 25, 498–509.
Bell, S., Morse, S., 2004. Experiences with sustainability indicators and stakeholder participation: a case study relating to a 'Blue Plan' project in Malta. Sustain. Dev. 12, 1–14.
Bostrom, A., Barke, R., Turaga, R.M.R., O'Connor, R.E., 2006. Environmental concerns and the new environmental paradigm in Bulgaria. J. Environ. Educ. 37, 25–40.
Christie, M., Hanley, N., Warren, J., Murphy, K., Wright, R., Hyde, T., 2005. Valuing the diversity of biodiversity. Ecol. Econ. 58, 304–317.
Chung, S.S., Poon, C.S., 2001. A comparison of waste-reduction practices and new environmental paradigm of urban and rural Chinese citizens. J. Environ. Manage. 62, 3–19.
Cleveland, C.J., Kaufmann, R.K., Stern, D.I., 2000. Aggregation and the role of energy in the economy. Ecol. Econ. 32, 301–317.
Sustainable urban development. In: Deakin, M., Mitchell, G., Nijkamp, P., Vreeker, R. (Eds.), The Environmental Assessment Methods, vol. 2. Routledge, Oxon.
DEFRA, 2005. Securing the Future: UK Government Sustainable Development Strategy. Sustainable Development Unit, Department for Environment, Food and Rural Affairs, London. Available at: http://www.sustainabledevelopment.gov.uk/publications/pdf/strategy/SecFut_complete.pdf.
De Marchi, B., Funtowicz, S.O., Lo Cascio, S., Munda, G., 2000. Combining participative and institutional approaches with multicriteria evaluation. An empirical study for water issues in Troina, Sicily. Ecol. Econ. 34, 267–282.
Dietz, T., Ostrom, E., Stern, P., 2003. The struggle to govern the commons. Science 302, 1907–1912.
Dunlap, R.E., van Liere, K.D., 1978. The new environmental paradigm: a proposed measuring instrument and preliminary results. J. Environ. Educ. 9, 10–19.
Dunlap, R.E., van Liere, K.D., Mertig, A.G., Jones, R.E., 2000. Measuring Endorsement of the new ecological paradigm: a revised NEP Scale – statistical data included. J. Soc. Issues 56, 425–442.
Dutta, M., Banerjee, B., Husain, Z., 2007. Untapped demand for heritage: a contingent valuation study of Prinsep Ghat, Calcutta. Tour Manage. 28, 83–95.
EC, 2008a. Indicative Guidance on ESF Evaluation Quality Standards (Draft). Evaluation Unit, DG EMPL, Brussels. Available at: www.ec.europa.eu/social/BlobServlet?docId=2301&langId=en.
EC, 2008b. Consolidated versions of the treaty on European Union and the treaty on the functioning of the European Union. Offic. J. Eur. Union C115, 1–388. Available at: http://eurlex.europa.eu/JOHtml.do?uri=OJ:C:2008:115:SOM:en:HTML.
EC, 2007. Directive 2007/60/EC on the assessment and management of flood risks. Offic J Eur Union L288, 27–34. Available at: http://eur703lex.europa.eu/LexUriServ/site/en/oj/2007/l_288/l_28820071106en00270034.pdf.
EC, 2003. Directive 2003/35/EC providing for public participation in respect of the drawing up of certain plans and programmes relating to the environment and amending with regard to public participation and access to justice Council Directives 85/337/EEC and 96/61/EC. Off. J. Eur. Union L156, 17–25. Available at: http://eur708lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:32003L0035:EN:HTML.
EC, 2001. Directive 2001/42/EC on the assessment of the effects of certain plans and programmes on the environment. Off. J. Eur. Union L197, 30–37. Available at: http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2001:197:0030:0037:EN:PDF.
EC, 2000. Directive 2000/60/EC establishing a framework for community action in the field of water policy. Off. J. Eur. Union L327, 1–73. Available at: http://eur714lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:32000L0060:EN:HTML.
EC, 1997. Council Directive 97/11/EC amending Directive 85/337/EE 715 C on the assessment of the effects of certain public and private projects on the environment. Off. J. Eur Union L073, 5–15. Available at: http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:31997L0011:EN:HTML.
Ecological Economics, 2002. The dynamics and value of ecosystem services: integrating economic and ecological perspectives. Ecol. Econ. 41, 367–567 (Special issue).
Farber, C.S., Costanza, R., Wilson, M.A., 2002. Economic and ecological concepts for valuing ecosystem services. Ecol. Econ. 41, 375–392.
Folke, C., Hahn, T., Olsson, P., Norberg, J., 2005. Adaptive governance of social-ecological systems. Annu. Rev. Environ. Resour. 30, 441–473.
Fosgerau, M., Bjorner, T.B., 2006. Joint models for noise annoyance and willingness to pay for road noise reduction. Transport Res. B Meth. 40, 164–178.
Funtowicz, S.O., Ravetz, J.R., 1993. Science for the post-normal age. Futures 25, 739–755.
Gamboa, G., Munda, G., 2007. The problem of windfarm location: a social multi-criteria evaluation framework. Energy Pol. 35, 1564–1583.
Gasparatos, A., El-Haram, M., Horner, M., 2008. A critical review of reductionist approaches for assessing the progress towards sustainability. Environ. Impact Assess. 28, 286–311.
Gasparatos, A., El-Haram, M., Horner, M., 2009. The argument against a reductionist approach for measuring sustainable development performance and the need for methodological pluralism. Account. Forum 33, 245–256.
Gibson, R., Hassan, S., Holtz, S., Tansey, J., Whitelaw, G., 2005. Sustainability Assessment: Criteria and Processes. Earthscan, London.
Glasson, J., Therivel, R., Chadwi, A., 2005. Introduction to Environmental Impact Assessment, third ed. Routldge, Oxon.
Gooch, G.D., 1995. Environmental beliefs and attitudes in Sweden and the Baltic states. Environ. Behav. 30, 520–534.
Habbani, K., Groot, W., Jelovac, I., 2006. Household health-seeking behaviour in Khartoum, Sudan: the willingness to pay for public health services if these services are of good quality. Health Pol. 75, 140–158.
Heinzerling, L., Ackerman, F., 2002. Pricing the Priceless: Cost–benefit Analysis of Environmental Protection. Georgetown Environmental Law and Policy Institute, Georgetown University, Washington D.C. Available at: http://www.law.george town.edu/gelpi/research_archive/cost_benefit_analysis/PricingThePriceless.pdf
Hitlin, S., Piliavin, J.A., 2004. Values: reviving a dormant concept. Ann. Rev. Soc. 30, 359–393.
HMSO, 1971. The Report of the Commission on the Third London Airport. HMSO, London.
Holden, E., Hoyer, K.G., 2005. The ecological footprint of fuels. Transp. Res. D Trans. E 10, 395–403.
Howarth, R.B., Wilson, M.A., 2006. A theoretical approach to deliberative valuation: aggregation by mutual consent. Land Econ. 82, 1–16.
van den Hove, S., 2006. Between consensus and compromise: acknowledging the negotiation dimension in participatory approaches. Land Use Pol. 23, 10–17.
Jacobs, M., 1997. Environmental valuation, deliberative democracy and public decision-making institutions. In: Foster, J. (Ed.), Valuing Nature? Economics, Ethics and Environment. Routledge, London, pp. 211–231.
Janssen, M.A., Ostrom, E., 2006. Empirically based agent-based models. Ecol. Soc. 11 (37) Available at: http://www.ecologyandsociety.org/vol11/iss2/art37/.
Jim, C.Y., Chen, W.Y., 2006. Recreation-amenity use and contingent valuation of urban greenspaces in Guangzhou, China. Landsc. Urban Plan. 75, 81–96.
Johannesson, M., Johansson, P.O., O'Connor, R., 1996. The value of private safety versus the value of public safety. J. Risk Uncertain 13, 263–275.
Kallis, G., Videira, N., Antunes, P., Guimaraes Pereira, A., Spash, C.L., CoccossisCorral Quintana, S., del Moral, L., Hatzilacou, D., Lobo, L., Mexa, A., Paneque, P., Pedregal Mateos, B., Santos, R., 2006. Participatory methods for water resources planning. Environ. Plann. C 24, 215–234.
Karp, D.G., 1996. Values and their effect on pro-environmental behavior. Environ. Behav. 28, 111–133.
Layard, R., 2005. Happiness: Lessons from a New Science. Penguin Books, London.
Leung, C., Jenni, R., 2002. Comparison of Chinese–Australian and Anglo-Australian environmental attitudes and behaviour. Soc. Behav. Person. 30, 251–262.

Lee, N., 2006. Bridging the gap between theory and practice in integrated assessment. Environ. Impact Assess. 26, 57–78.
Lefroy, E., Rydberg, T., 2003. Emergy evaluation of three cropping systems in Southwestern Australia. Ecol. Model. 161, 195–211.
Lockwood, M., 1998. Integrated value assessment using paired comparisons. Ecol. Econ. 25, 73–87.
Mackay, K., 2007. How to Build M&E Systems to Support Better Government. World Bank, Washington DC. Available at: http://www.worldbank.org/ieg/ecd/docs/How_to_build_ME_gov.pdf.
Madlener, R., Stagl, S., 2005. Sustainability-guided promotion of renewable electricity generation. Ecol. Econ. 53, 147–167.
Mataria, A., Giacaman, R., Khatib, R., Moatti, J.P., 2006. Impoverishment and patients' "willingness" and "ability" to pay for improving the quality of health care in Palestine: an assessment using the contingent valuation method. Health Pol. 75, 312–328.
Messner, F., Zwirner, O., Karkuschke, M., 2006. Participation in multi-criteria decision support for the resolution of a water allocation problem in the Spree River basin. Land Use Pol. 23, 63–75.
Milfont, T.L., Duckitt, J., Cameron, L.D., 2006. A cross-cultural study of environmental motive concerns and their implication for pro-environmental behavior. Environ. Behav. 38, 745–767.
Munda, G., 2006. Social multi-criteria evaluation for urban sustainability policies. Land Use Pol. 23, 86–94.
Munda, G., 1996. Cost–benefit analysis in integrated environmental assessment: some methodological issues. Ecol. Econ. 19, 157–168.
Munda, G., Nardo, M., 2005. Constructing Consistent Composite Indicators: The Issue of Weights (EUR 21834 EN). Institute for the Protection and Security of the Citizen, Ispra. Available at: http://crell.jrc.ec.europa.eu/Well-being/papers/Munda%20Nardo%20euroreport1.pdf.
Ness, B., Urbel-Piirsalu, E., Anderberg, S., Olsson, L., 2007. Categorising tools for sustainability assessment. Ecol. Econ. 60, 498–508.
Neumayer, E., 2004. Weak Versus Strong Sustainability: Exploring the Limits of Two Opposing Paradigms, second ed. Edward Elgar, Cheltenham.
Niemeyer, S., 2005. Deliberation in the Wilderness: The North Queensland Citizen Jury. Land & Water Australia, Canberra. Available at: http://downloads.lwa2.com/downloads/publications_pdf/PR040804.pdf.
Niemeyer, S., Spash, C., 2001. Environmental valuation analysis, public deliberation, and their pragmatic syntheses: a critical appraisal. Environ. Plann. C. 19, 567–585.
OECD, 2008. Handbook on Constructing Composite Indicators: Methodology and User Guide. OECD publication Code: 302008251E1 Available at: http://compositeindicators.jrc.ec.europa.eu/Handbook.htm.
OECD, 2007. DAC Evaluation Quality Standards (for Test Phase Application). Organisation for Economic Cooperation and Development, Paris. Available at: http://www.oecd.org/dataoecd/34/21/38686856.pdf.
OECD, 2003. Composite Indicators of Country Performance: a Critical Assessment. STI Working Paper 2003/16. Organisation for Economic Cooperation and Development, Paris. Available at: http://www.itu.int/osg/spu/ni/wsisbridges/linked_docs/Background_papers/otherdocs/OECD_WP_2003_16.pdf.
OECD, 1998. Review of the DAC Principles for Evaluation of Development Assistance. Organisation for Economic Cooperation and Development, Paris. Available at: http://www.oecd.org/dataoecd/63/50/2065863.pdf.
Ojea, E., Loureiro, M.L., 2007. Altruistic, egoistic and biospheric values in willingness to pay (WTP) for wildlife. Ecol. Econ. 63, 807–814.
Patterson, M., 1998. Commensuration and theories of values in ecological economics. Ecol. Econ. 25, 105–125.
Pearce, D., 1993. Blueprint 3. Measuring Sustainable Development. Earthscan Publications Ltd., London.
Pope, J., Annandale, D., Morrison-Saunders, A., 2004. Conceptualising sustainability assessment. Environ. Impact Assess. 24, 595–616.
Rauwald, K.S., Moore, C.F., 2002. Environmental attitudes as predictors of policy support across three countries. Environ. Behav. 34, 709–739.
Sagoff, M., 1998. Aggregation and deliberation in valuing environmental public goods: a look beyond contingent pricing. Ecol. Econ. 24, 213–230.
Salgado, P.P., Quintana, C.S., Guimarães Pereira, A., del Moral Ituarte, L., Pedregal Mateos, B., 2009. Participative multi-criteria analysis for the evaluation of water governance alternatives. A case in the Costa del Sol (Málaga). Ecol. Econ. 68, 990–1005.
Schultz, W.P., Zelezny, L.C., 1998. Values and pro-environmental behavior: a five country survey. J. Cross Cul. Psychol. 29, 540–558.
Schultz, W.P., Zelezny, L.C., 1999. Values as predictors of environmental attitudes: evidence for consistency across 14 countries. J. Environ. Psychol. 19, 255–265.
Schwartz, S.H., 1992. Universals in the content and structure of values: theoretical advances and empirical tests in 20 countries. Adv. Exp. Soc. Psychol. 25, 1–65.
Sciubba, E., 2003. Extended exergy accounting applied to energy recovery from waste: the concept of total recycling. Energy 28, 1315–1334.
Stagl, S., 2007. Emerging Methods for Sustainability Valuation 813 and Appraisal. Sustainable Development Research Network, London. Available at: http://www.sd-research.org.uk/post.php?p=129.
Stern, P.C., 2000. Towards a coherent theory of environmentally significant behavior. J. Soc. Issues 56, 407–424.
Stern, P.C., Dietz, T., 1994. The value basis of environmental concern. J. Soc. Issues 50, 65–84.
Stern, P.C., Dietz, T., Kalof, L., 1993. Value orientations, gender and environmental concern. Environ. Behav. 25, 322–348.
Stern, P.C., Dietz, T., Kalof, L., Guagnano, G.A., 1995. Values, beliefs and pro-environmental action: attitude formation towards emergent attitude objects. J. Appl. Soc. Psychol. 25, 1611–1636.
Stern, P.C., Dietz, T., Abel, T., Guagnano, G.A., Kalof, L., 1999. A value-belief-norm theory of support for environmental movements: the case of environmental concern. Hum. Ecol. Rev. 6, 81–97.
Spash, C., 2000. Ecosystems, contingent valuation and ethics: the case of wetland recreation. Ecol. Econ. 34, 195–215.
Spash, C., 2006. Non-economic motivation for contingent values: rights and attitudinal beliefs in the willingness to pay for environmental improvements. Land Econ. 82, 602–622.
Spash, C., 2007a. Deliberative monetary valuation (DMV): issues in combining economic and political processes to value environmental change. Ecol. Econ. 63, 690–699.
Spash, C., 2007b. Deliberative Monetary Valuation (DMV) in Practice. Socio-Economics and the Environment in Discussion. CSIRO Working Paper Series 2007-04. CSIRO, Canberra. Available at: http://www.csiro.au/files/files/pewk.pdf.
Spash, 2008. Deliberative monetary valuation and the evidence for a new value theory. Land Econ. 84, 469–488.
Spash, C., Hanley, N., 1995. Preferences, information and biodiversity preservation. Ecol. Econ. 12, 191–208.
Spash, C., Urama, K., Burton, R., Kenyon, W., Shannon, P., Hill, G., 2009. Motives behind willingness to pay for improving biodiversity in water ecosystems: economics, ethics and social psychology. Ecol. Econ. 68, 955–964.
Stair, A., Rephann, T.J., Heberling, M., 2005. Demand for public education: evidence from a rural school district. Econ. Educ. Rev. 25, 521–531.
Thompson, E., Berger, M., Blomquist, G., Allen, S., 2002. Valuing the arts: a contingent valuation approach. J. Cult. Econ. 26, 87–113.
Ton, S., Odum, H.T., Delfino, J.J., 1998. Ecological–economic evaluation of wetland management alternatives. Ecol. Eng. 11, 291–302.
Tyrvainen, L., 2001. Economic valuation of urban forest benefits in Finland. J. Environ. Manag. 62, 75–92.
UN, 1992. Agenda 21. United Nations Department of Economic and Social Affairs, Division for Sustainable Development, New York. Available at: http://www.un.org/esa/sustdev/documents/agenda21/english/agenda21toc.htm.
UNECE, 2007. Resource Manual to Support Application of the Protocol on SEA. United Nations Economic Commission for Europe, Geneva. Available at: http://www.unece.org/env/eia/sea_manual/welcome.html.
Vatn, 2009. An institutional analysis of methods for environmental appraisal. Ecol. Econ. 68, 2207–2215.
Vatn, A., 2005a. Rationality, institutions and environmental policy. Ecol. Econ. 55, 203–217.
Vatn, A., 2005b. Institutions and the Environment. Edward Elgar, Cheltenham.
Wallis, A.M., 2006. Sustainability indicators: is there consensus among stakeholders? Int. J. Environ. Sustain. Dev. 5, 287–296.
Wallis, A.M., Richards, A., O'Toole, K., Mitchell, B., 2007. Measuring regional sustainability: lessons to be learned. Int. J. Environ. Sustain. Dev. 6, 193–207.
WCED, 1987. Our Common Future. Oxford University Press, Oxford. Available at: http://www.worldinbalance.net/agreements/1987-brundtland.html.
Widmer, T., 2005. The development and status of evaluation standards in western Europe. New Persp. Eval. 104, 31–42.
Wilson, M.A., Howarth, R.B., 2002. Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation. Ecol. Econ. 41, 431–443.
Winkler, R., 2006. Valuation of ecosystem goods and services. Part 1: an integrated dynamic approach. Ecol. Econ. 59, 82–93.
World Bank, 1996. The World Bank Participation Sourcebook. World Bank, Washington DC. Available at: http://www.worldbank.org/wbi/sourcebook/sbhome.htm.
World Bank, 2002. Sleeping On Our Own Mats: An Introductory Guide to Community-based Monitoring and Evaluation. World Bank, Washington DC. Available at: http://siteresources.worldbank.org/INTPCENG/2145741116505633693/20509339/communitybased.pdf.
World Bank, 2004. Monitoring and Evaluation: Some Tools, Methods and Approaches. World Bank, Washington DC. Available at: http://www.worldbank.org/ieg/ecd/tools/.
Yang, H., Li, Y., Shen, J., Hu, S., 2003. Evaluating waste treatment, recycle and reuse in industrial system: an application of the emergy approach. Ecol. Model. 160, 13–21.
Zmeureanu, R., Wu, X.Y., 2007. Energy and exergy performance of residential heating systems with separate mechanical ventilation. Energy 32, 187–195.