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Transferring environmental value estimates: Issues and alternatives

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ABSTRACT

Environmental value transfer needs to be understood in the context of scientific information use in general. This provides a different perspective upon the reasons why benefit transfer in particular appears so controversial and raises concerns over the limited types of validity testing being undertaken by those supporting such applications as ecosystem services valuation. Another key issue, which we emphasise, is the unintentional challenge to standard economic theory raised by the models used to conduct value transfers. Existing value transfer practice reveals the need for a more inclusive approach if environmental values are to be addressed. We argue that there are robust alternative means for including multiple environmental values in decision processes, these cannot be dismissed out of hand, and analysts should be expanding their understanding of the available approaches which include attitude and norm measures, multi-criteria analysis and participatory deliberative institutions.

1. Introduction

Monetary valuation of the environment was developed within the bounds of neoclassical microeconomic theory with the main application intended to be small scale projects. This project focus was meant to avoid large income effects (i.e., changing the marginal utility of money) and avoid violations of ceteris paribus (i.e., changing other prices). However, in practice environmental cost–benefit analysis (CBA) has been applied in a variety of other contexts and often been regarded as a policy aid rather than a project appraisal tool. At the same time analysis has moved from valuing site specific recreation to global ecosystems functions and their associated "services"[1]. Neoclassically trained micro-economists are rightly concerned over the relationship of the values being produced to any theoretical understanding. Indeed such values are often justified on the basis of pragmatism, i.e. supplying dollar values in a world run by business and where the "Treasury Department" is believed to be the strongest arm of the government. In this context the transfer of monetary value estimates from an original primary study to another time and place, where ceteris paribus is ignored, is regarded as a cost saving and fast way in which to supply "information" to politicians and administrators that the environment is worth something.

Of the two main approaches function transfer is regarded as more robust because it uses a set of explanatory variables upon which values are deemed to depend, while unit transfer tends to be a more simple adoption of monetary numbers out of one

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[1] The 'ecosystem service' literature discusses a range of 'uses' which are implicit in human activity and therefore only indirectly 'used'; this encounters problems similar to indirect/passive use values. The inclusion of functions essential to life implies considerable confusion as to what exactly is the meaning, if any, of the money numbers produced from this literature, i.e. trade price, capital value, marginal WTP or perhaps WTA, or just a convenient number for political purposes?

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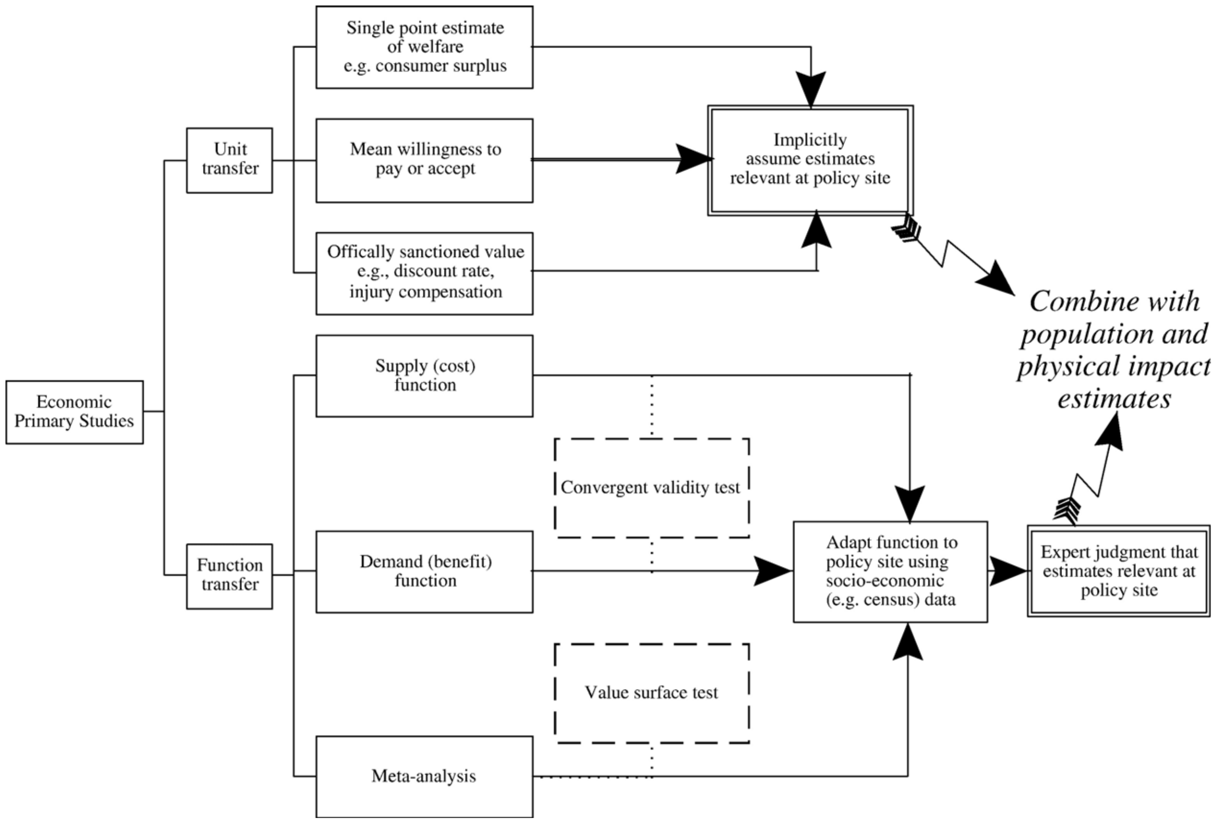

Fig. 1 – Value transfer approaches.

context and into another. These supposedly quick and cheap alternatives have proven controversial in the specific context of valuing the benefits from environmental improvements, and the losses from natural resource damages. Concerns include the quality of the available data and the poor conduct of primary studies (Green, 2004), and have stimulated calls for different methods (Morrison et al., 2002), and so by implication the need for more primary studies. However, Smith et al. (2002: 135) claim that resource constraints mean "…the choice is not between a new study or a transfer but between transfer and qualitative judgment", i.e. new studies are not on the agenda. Yet even such supporters openly admit that a decade of research evaluating value transfer studies shows "the conventional approaches are very unreliable" (Smith et al., 2002: 134).

The empirical validation of results, as in the scientific experimental approach, is advocated by mainstream economic methodology. Capital costs provide a good example, of regular value transfer, which can be checked ex post by actual market prices. Thus, Green (2004:1) is able to cite errors as ranging from 30% for routine coastal and flood defence projects up to 3000% for unique, cutting edge technologies. In the absence of market prices error estimates are calculated by comparing transferred values with hypothetical on-site willingness to pay (WTP) or accept (WTA) studies. Errors in spatial transfer of mean WTP have led to environmental value transfer being recommended only where the demand for accuracy is relatively low (Navrud and Bergland, 2001 p.12). During the workshop from which this special issue arose, March 2005 in Washington DC, errors in the order of 750% were cited for idealised spatial transfer tests (i.e., ignoring distortions due to time). In the proceedings of that meeting, Rosenberger (2005) reported errors up to 577% for unit transfer, 475% for function transfer, and 7028% for meta-analysis. There is then a debate over the technical approach by which numbers might be "calibrated" to achieve "theoretical consistency".

Brouwer (2000: 146–147) contests the ability of technical restrictions to aid practical environmental value transfer and sees this approach as "part of the problem, not the solution". He regards economics as having hit an operational boundary with respect to comprehending the underlying motives for environmental values. He specifically identifies the importance of qualitative factors and advocates deliberative participation of stakeholders. The real concern is then over the extent to which "qualitative judgment" is required and permitted in policy processes. The implicit model of mainstream economists places quantification at the forefront with experts producing objectivity via monetary numbers. Yet such economic analysts admit this information is only one aspect of input to a political decision process which they regard as a black box. This rather begs the question as to why advocates, who claim little need to understand the policy process, place so much faith in highly uncertain transferred monetary numbers on the grounds of pragmatism? The argument that there are no alternatives is false (see Spash et al., 2004 and examples in that edited volume). One aim here is to indicate some of the alternatives now available and why they are needed.

We start by placing value transfer within the context of information transfer in the natural and social sciences. This raises the question as to how value transfer can establish valid

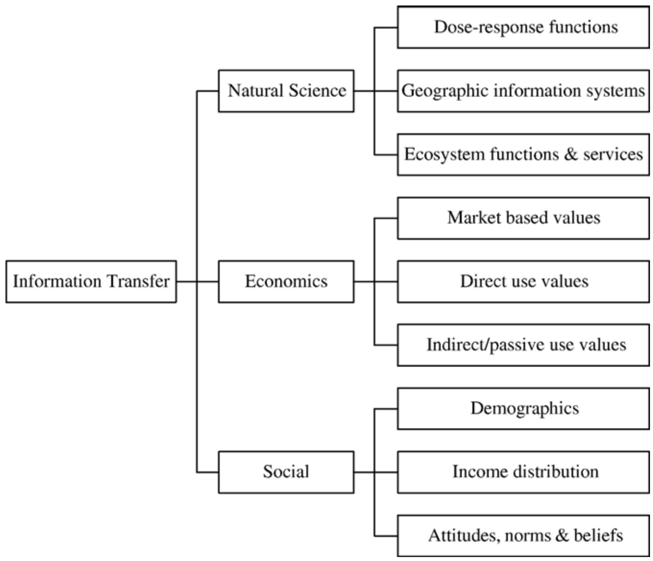

Fig. 2 – Range and sources of information use in environmental valuation.

results given the unobservable nature of most environmental values of current interest. The discussion of validity indicates some of the more fundamental theoretical issues which need to be addressed. We highlight the role of a wide range of socio-economic and demographic variables in analysing environmental values, and the implications of broadening understanding of influences on human behaviour for choice theory. After explaining and over-viewing these issues a range of alternative approaches for achieving the same ends as value transfer are discussed.

2. Placing value transfer in context

All monetary value transfers require primary original studies upon which values are based and justified. Fig. 1 summarises the economic approaches by which value transfer attempts to take such primary data to produce and validate monetary estimates of environmental values. There are two main categories of monetary value transfer: the unit and function transfer.

Under unit transfer three basic approaches supply the numbers. The simple point estimate defines a unit value of welfare, such as an average amount for consumer surplus per fishing trip under specific water quality conditions (Smith et al., 2002: 134). Hedonic pricing, the travel cost model, the contingent valuation method (CVM) and choice experiments can all provide WTP amounts.[2] Alternatively, the transfer of values can occur via officially sanctioned numbers, and perhaps the most common designated unit value is the trade price over time, i.e. the discount rate (for a comprehensive assessment see Price, 1993). Standard values are also found for loss of life, such as recommended in the UK for government transport projects. That standard values are designated fails to prevent considerable controversy over the use of both discounting and statistical life calculations (as discussed in Chapters 7 and 8 of Spash, 2002). Unlike function transfer, the unit transfer approach commonly takes numbers at face value and thus implicitly assumes they are relevant to a specific project/policy context, without any testing.

Function transfer is generally regarded as the more appropriate and theoretically rigorous approach in the value transfer literature (Brouwer, 2000; Chattopadhyay, 2003). The function aims to explain economic costs or benefits (mainly WTP) in terms of a set of explanatory variables. Meta-analysis can be used to combine functions from several studies, the original data usually being unavailable, and assumes there is an underlying meta function linking the valued object/activity and socio-economic characteristics across space and time. Functions can be tested for statistical rigour and adapted to the policy site. The latter requires policy site information on the relevant socio-economic variables in the value function, although this data may be absent raising issues of interpolation.

The concentration in the value transfer literature has been upon the economic aspects, as shown in Fig. 1, although social, economic and natural science information are necessary. A lack of knowledge in any of these dimensions can lead practitioners to borrow information from another context. Information transfer is then the overarching concept relative to which value transfer is but a sub-category. Fig. 2 exemplifies the type of data typically being transferred when an economic analysis of environmental change is undertaken, and highlights the categories of information being employed. In terms of value transfer the question to be addressed is why it should be a distinct area of controversy compared to the transfer of other data?

2.1. *Natural science information transfer*

The reliability of natural science data is generally unquestioned in economic analysis of environmental change. Rarely is an economic study conducted in association with a new piece of scientific research or are site specific current damage estimates obtained. Instead information is commonly transferred. Let us consider each information source in Fig. 2.

Dose-response functions are widely employed in environmental toxicology (Spash and McNally, 2001). For example, the benefits of reducing tropospheric ozone pollution can involve increased agricultural crop yields leading to welfare improvements for producers and consumers estimated via economic models (Spash, 1997a). The physical relationship between ambient air quality and these impacts is taken from experimentally derived dose-response functions; these are transferred across time, space and plant species, as well as being extrapolated to different ambient air quality conditions. In theory, the repeatability of controlled experiments should, via observation, define errors and probabilities for outcomes. Empirical testing of hypotheses relating to a physically objective state of the world is the underlying methodological defence of validity within the natural sciences. The knowledge gained falls within the area of weak uncertainty and normal science as opposed to strong

[2] Willingness to accept is neglected despite being the most reasonable approach for assessing environmental damages (Knetsch, 1994, 2005). Mean WTP is the most commonly reported and extracted value while the CVM is the most prevalent source of economic value estimates for environmental changes.

uncertainty and post normal science (see Funtowicz and Ravetz, 1993, 1994; Spash, 2002). However, the use of data out of context and with excessive aggregation can be problematic despite a well founded experimentally derived set of base information. For example, estimating crop responses to climate change under the enhanced Greenhouse Effect shows the inadequacies of existing scientific knowledge (Erickson, 1993), although this has done little to prevent its application in economic studies (Spash, 2002).

In other areas scientific information is being transferred with less well known but suspect consequences. The use of geographic information systems with travel cost models (e.g. Brainard et al., 1999) is regarded as a technical approach, avoiding the CVM controversy. However, site characteristics are valued individually and out of context to achieve transfer across sites. As Brouwer (2000: 146) notes, individual characteristics can be highly interdependent as perceived from either a social or natural science perspective making the transferred values open to dispute.

Such interdependence is also typically neglected when supplying monetary numbers for 'natural capital'. Ecosystems functions have been categorised as service types such as regulation, habitat provision, production and information (e.g. Batker et al., 2005) and some ecologists have defined separate sub-categories of 'goods and services' (e.g., de Groot et al., 2002). In effect, such ecologists use their expert opinion to transfer knowledge about the likely role and importance of functions. Services ranging from gas regulation to "spiritual and historic information" provision are related to physical characteristics which are then regarded as universal. This type of practice moves well beyond the standard context within which natural science normally operates.

Thus, a range of natural science information is being transferred with applications ranging from the theoretically well established experimental to the contentious and more recent, ecologically driven, systems approaches. This appears similar to the move from areas of normal sciences to those of post-normal science and from weak to strong uncertainty. While scientific information transfer is a common and, indeed normally, essential part of environmental CBA, this goes largely unnoticed and rarely noted. The apparent disregard for the validity of the scientific information employed is in stark contrast with the transfer of values which themselves result from the use of such scientific information in combination with economic methods.

2.2. *Social and economic information transfer*

The most commonly discussed and disputed area of information transfer in environmental economics is that relating to the monetary benefits associated with environmental changes. Benefit transfer is in fact a sub-category of value transfer because both costs and benefits are in practice transferred. Few authors acknowledge the common use of cost transfers. In addition, less certainty is attributed to benefits even though what constitutes a benefit or a cost is relative to the policy/project on–off scenario, and cost and benefit categories can interchange on that basis (Spash, 1997b). Fig. 2 shows economic information is being sought on both direct and indirect use values.

Direct use value transfer requires socio-economic data for the policy site in order to avoid errors in terms of population demand and valuation, see also Fig. 1. For example, consider two sites judged to have similar physical characteristics, but where A has a few people who live locally and are prepared to pay a large amount while B has an unknown set of population characteristics which could include: a lot of people being prepared to pay a small amount who live locally, a few people prepared to pay a large amount who live far away, or some in between combination. In order to undertake the transfer of recreational values, information on population characteristics affecting use and unit value are required (Green, 2004: 5–6). However, collection of new socio-economic data violates the claimed advantages of such transfers namely, simplicity and avoidance of primary data collection.

The problem is worse for indirect/passive use values[3], because there is no associated observable behaviour and only stated preference methods can be employed to assess the value. There is no easy approach to identifying the relevant population of those valuing iconic habitats or species, which are known across the globe. We might believe that World Heritage Sites, for example, are valued by the entire human population, but who is prepared to pay for them and why? If species are lost, who should be potentially compensated to meet the Kaldor–Hicks criterion? Endangered species cannot be valued outside the context of their habitats, which means entire ecosystems existence needs to be considered if scenarios are really to address existence value. The categories of bequest and existence value for entire ecosystems may be considerable (e.g. Amazonian rain forest), or if some small obscure place perhaps miniscule (a Scottish bog in the highlands). Species can also be highly valued and rare in one location and common in another or even regarded as a pest to be eradicated (e.g. possums in New Zealand as opposed to Australia).

The difference between socio-economic information transfer and that in the natural sciences is the move away from a physical reality which can be observed within a set of controlled circumstances. There is an evident conflict between regarding the world as a physically objective reality and a subjective complex human system. From the modern economists' perspective values are based upon the preferences of individuals, but from the ecologists' perspective such things as ecosystems existence and the value of bequesting ecosystems to future generations are defined by the physical characteristics of that ecosystem. There are then two incongruous classification systems: one arising from environmental economics (primarily the CVM), and the other arising from ecology and ecosystems functions. Yet the literature valuing ecosystem services 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. The problem facing those ecologists promoting ecosystems services valuation is that most of what they deem valuable is unlikely to produce meaningful WTP amongst the general

[3] Indirect/passive use values are sometimes incorrectly termed non-use values; there are no non-use values in economics because all economic value derives from the utility its provides humans. The categories of indirect/passive use value are defined in the literature as option, bequest and existence. However, any list of values is contentious and appears somewhat arbitrary, especially if it claims to be comprehensive e.g. total economic value.

population e.g., under a CVM scenario[4]. Value transfer is then being used to circumvent this basic issue. This of course raises the question as to how any value transfer can claim validity?

3. Value transfer validity

In general the error in value transfer is expected to be smaller where the correspondence is close between the set of factors at the study and policy sites (Rosenberger, 2005). Specific conditions of similarity can be gleaned from the literature and we found low errors are expected when the following are matched at the two sites:

(i) the environmental good/service, its quantity/quality and the change in quantity/quality;
(ii) the population, their use of the good/service and their characteristics;
(iii) constructed market characteristics;
(iv) institutional setting
(v) time between primary collection and transfer;
(vi) geographical location.

However, very similar valuation scenarios, as used in experimental tests of convergent validity, are uncommon in practice (Barton, 2002).

While convergence (as highlighted in Fig. 1) has been the primary test in the literature, face, construct, predictive, criterion and divergent validity are also relevant (Green, 2004). Face validity is whether results are intuitively plausible; construct validity concerns consistency with theoretical foundations; predictive validity is whether the expected outcome was matched by the actual outcome; criterion validity relates to corroborating factors which confirm model prediction and is important where predictions cannot be confirmed by direct observation; convergent validity requires different techniques to give the same results; divergent validity requires the same technique to give different results where context predicts that should occur, e.g. the CVM measuring WTP for two different population income distributions. Scientific reliance upon predictive validity is impossible for value estimates of non-market goods and services where there are, by definition, no observed market prices ex post. This increases the importance which should be paid to the full range of validity tests.

In practice different aspects of validity seem to be given little attention. For example, one critique of the value transfer studies relating to "ecosystem services" is that they fail both construct and face validity tests. That some calculations of an annual value for the Earth's primary ecosystems exceed the income available to pay such an amount violates income constraints (Bockstael et al., 2000). Such values therefore appear both intuitively unbelievable and theoretically inconsistent.

In all value transfer applications the defensibility of the figures will be the ultimate test. If the stakes are high then the political liability of using uncertain numbers will also be high. Indeed in areas of application where figures are most hotly contested, such as compensation payments for environmental damages (e.g., the Exxon Valdez case, see Hausman, 1993), the original primary data is already contentious. That the quality of primary studies determines the quality and applicability of data for transfer is too easily neglected when the emphasis is placed on producing a number on grounds of pragmatism. For example, many studies under the CVM, and especially earlier studies, were conducted for research purposes using non-representative convenience samples (e.g., undergraduate students) and performed by untrained interviewers (e.g. postgraduate students) without any quality control. Key areas where primary studies have proven inadequate in the past include:

(i) Survey design: e.g., easily understood and pre-tested language, taking onboard all feedback from focus groups not just that which is convenient;
(ii) Data collection e.g., sample size, collection methods, sample representation of the general population, randomised selection;
(iii) Economic methods: basis for the approach in a theoretical model;
(iv) Empirical techniques: correct statistical tests;
(v) Explanatory power: being very low;
(vi) Reliability and validity tests: regression results explaining WTP as a function of relevant factors.

While primary studies prove inadequate to the job, the attempts to transfer them have also revealed that environmental values are poorly defined in economic terms. That is, values are found which represent social and moral commitments of a non-consequentialist and non-utilitarian kind, and the context within which values arise is highly relevant to their expression.

4. Are value transfer practices consistent with economic theory?

Monetary valuation, with or without value transfer, has been confronted by a series of fundamental questions. These include concerns over whether values are commensurable (e.g., Chang, 1997; O'Neill, 1997; Martinez-Alier et al., 1998; Aldred, 2002); aggregation of values across individuals is possible or defensible, questioning the Kaldor–Hicks criterion; people have preferences for passive/indirect use values (McFadden and Leonard, 1993; Diamond and Hausman, 1994); and the observed divergence of WTP and WTA measures far beyond what can be reasonably explained by economic theory (e.g., Knetsch, 2005). Such issues are of crucial importance for the choice of valuation methods, and the role various estimates should play in policy processes.

In the case of Kaldor–Hicks, or the so-called Potential Pareto Improvement rule, welfare economics accepts that some should lose if the gain by others is large enough to potentially compensate the losers. Being at the core of CBA, this rule raises serious normative problems. Moreover, if environmental choices are about common goods, the evaluation of arguments may be a

[4] The feasibility of a realistic payment scenario seems doubtful; studies on ecosystems services tend to involve changes in a large number of functions (e.g. Batker et al., 2005 referring to twenty three diverse functions).

better basis for social choice than individual WTP. Concerning WTP and WTA we observe that WTP is dominantly used without seriously discussing the implied rights assumptions. The problem might be less serious if WTP were fairly equal to WTA bids, rather than often 3 to 4 times smaller. In addition, as Knetsch (2005) has explained, equality between WTP and WTA would be counter to behavioural evidence showing that individual's value gaining something very differently than losing that exact same thing. In effect use of WTP in a context where WTA is theoretically appropriate substantially alters the value estimated. Value transfer may exacerbate such problems — e.g., the perception of rights may be different at the policy site as compared with the study site, making number transfer fundamentally inconsistent. Still, the above issues are all general to CBA, i.e. they exist regardless of whether values are transferred, and given our focus here they are excluded from further discussion.[5] Instead we focus upon a fundamental issue specific to value transfer namely explaining the motives behind values and their variations.

In order to understand whether a study is successful, there must be an understanding of the theoretical reasons why values should diverge. Brouwer (2000: 136) notes that "…, even if the goods, the sites where they are found and their user groups are similar, the benefits derived from these goods are not necessarily the same if the distribution of the population and their characteristics around the sites are not the same (Loomis, 1992)." Primary data must then be available if adjustments for these factors are to be made between the study and policy sites. Demographic data availability may be limited and its quality in many countries may be questionable, which will affect the practical application of value transfer (Brouwer, 2000: 141). However, non-demographic data are even more troublesome because they are normally unavailable and therefore would require collection via primary survey work. More important than these practical issues are the theoretical implications.

Micro-economic theory, which underpins economic valuation and thus value transfer, is based on methodological individualism and hence regards preferences as strictly individual. They are meant to remain uninfluenced by social or institutional circumstances. This is a core methodological foundation of the discipline (Becker, 1976; Stigler and Becker, 1977). However, reasons for this position go beyond methodological justifications because the whole idea of consumer sovereignty, so crucial for modern welfare theory, hinges upon the assumption (Vatn, 2005). What is then of interest are the practice and findings from value transfer studies.

Most studies that are based upon function transfer include a limited set of socio-economic and demographic variables which typically cover income, age, gender, and education. Such variables might be argued to have a clear economic meaning in that the same person would prefer different things if they changed status according to these dimensions, without implying that there are any necessary changes taking place in the basic preference function. Variation across sites concerning income — both distribution and average income levels — would be expected to change WTP/WTA for environmental goods. Including income is hence consistent with the model. Including age may also be quite easily defended by assuming a person prefers different things when old, rather than young, due to their different physical status. No social influence on preferences has necessarily occurred through their lifetime. In the case of gender Barton (2002) emphasizes that it's inclusion is rather ad hoc in terms of the economic model. Despite this, gender is commonly found in bid functions; for example, a common expectation is that women are more pro-environmental and therefore have a higher WTP. Many economist seem unaware that gender is a social construct through which work, rights, responsibilities and relationships are organised (Green, 2004). Moving next to the case of education, a claim that no social process influences preferences in this context is simply unbelievable. Thus, even this limited set of typical variables appears inconsistent with economic theory. Certainly, where a wider range of variables is included problems of consistency are even greater.

Studies like those by Bateman et al. (2005), Brainard, Lovett, and Bateman (1999), and Chattopadhyay (2003) include factors such as ethnicity, socio-economic group, social class identity and neighbourhood characteristics.[6] Non-demographic factors include environmental attitudes and other motives behind behaviour. Culture and a sense of place have been mentioned as potentially key factors (Brouwer, 2000). For example, a whale would be expected to be a different entity in Japan and Norway as opposed to Western Europe and North America where it is more commonly regarded as a sentient being rather than an item of food. Yet the same point applies to local environments where specific social and cultural values are intertwined with people's identity and family history.

The literature on value transfer also reveals quite different conclusions as to the effect of including socio-economic and demographic variables. Some studies conclude that the standard variables used are often insignificant or capture only a small part of the total variation (Brouwer, 2000; Barton, 2002; Shrestha and Loomis, 2003; Jiang et al., 2005). Some, like Rosenberger (2005), emphasize that results are generally better the more socially and culturally alike are the study and policy sites. Ready et al. (2004), in their study of transfers between countries, cite cultural similarity as explaining why a function transfer with socio-economic/demographic variables did not improve results compared to a simpler unit transfer. There are also studies showing that the same socio-economic/demographic conditions produce significantly different stated WTP results (Brouwer, 2000; Rozan, 2004). Finally, some recognize that including socio-economic variables relating to attitudes increases the validity of results (e.g., Brouwer, 2000; Barton, 2002; Jiang et al., 2005).

Indeed, the importance of attitudes in understanding WTP has been repeatedly noted (Brouwer, 2000: 141; Barton, 2002: 161;

[5] For our position on various issues in CBA see Spash (2002), Spash and O'Neill (2000), Vatn (2004, 2005), and Vatn and Bromley (1994). The remit given to the authors in the current paper was to address value transfer and avoid a general review of problems in CBA.

[6] These studies all focus strongly on the spatial aspect — often using GIS — bringing the analyses in contact with data showing spatial variation in socio-demographics, indicating that local cultures exist and are influential. Note, Morrison et al. (2002) conclude transferring value estimates between sites is easier than between populations.

Chattopadhyay, 2003: 579; Green, 2004: 9) but given little depth of research. Even when a key aspect of a transfer study focuses upon attitudinal scales (Jiang, Swallow, and Mcgonagle, 2005) their construction, measurement, testing and theoretical basis remain largely unexplained. This is important for value transfer because ignoring such a primary factor will prove misleading. For example, pro-environmental attitudes are associated with rights and lead to higher WTP (Kotchen and Reiling, 2000; Spash, 2000a, in press). Thus, measuring environmental attitudes directly is important because there is no reason to expect say gender to capture this factor, e.g. women with other standard factors equal in New York to act identically to women in Seattle or those in the UK to match those in Italy or the USA. A more controversial proposition is that CVM and choice experiments are actually measuring attitudes (i.e. like and dislike) with respect to an object rather than preferences (Kahneman and Sugden, 2005). Yet, any variable showing high correlation with WTP, which improves the bid function, is more likely to be regarded as a positive factor in economic analysis due to the neglect of underlying meaning.

Thus, potential inconsistencies can be identified in relation to the underpinning economic model, but discussion about why certain variables are included is mostly absent from value transfer studies. Apparently obtaining (potentially) better value transfer estimates is taken to legitimise less theoretical stringency and rigour. Standard socio-economic/demographic variables seem to have restricted explanatory power; an observation also supported by the fact that results may vary substantially despite the status of variables being similar. There is then a clearly identified need for understanding how individual and social factors interact. While including a wider range of variables increases costs when undertaking a value transfer (field data must also be gathered at the policy site) it supports the idea that preference formation is a complex process. The results may be insightful for validating value transfer, but more important is the potential to help the economics profession establish a better understanding of what motivates choice. This is an urgent task which should lead to the development of better approaches for including environmental values and greater appreciation of various alternatives for aiding decision processes.

5. Alternative means of making choices

There are three broad groupings of approaches to aiding decision processes which can either complement or replace value transfer depending upon circumstances. First are measures of motives underlying human behaviour which have developed quantitative scales for analysis and prediction. Second are multiple criteria analyses (MCA) which place economic analyses in the context of other decision variables. Third are the range of approaches aiming to involve stakeholders and/or the general public in deliberative participatory events.

Environmental valuation, and particularly the CVM, has raised a series of concerns over what motivates individuals to state an intention to pay for an environmental improvement (Svedsater, 2003). There is now far greater acceptance by economists that psychological motives are important and preferences are often constructed in response to research aiming to discover how people value the environment. Kahneman and Sugden (2005) highlight several reasons why monetary values derived from stated preference approaches can diverge from a measure of utility based upon actual experience evaluated ex post. This leads to the need for evaluating subjective well-being or happiness directly rather than relying upon the CVM, and measurement of experience utility is described as a realistic option. At the same time, Kahneman and Sugden (2005) note their differences on the way forward with Kahneman seeing measurement of experience utility as one component of useful information on social good, while Sugden opts for measures of preference satisfaction. Either way the call upon social psychology to aid economic understanding of human behaviour is important.

Due to the standard economic acceptance of preferences at face value, the motives behind preferences have tended to be weakly analysed. In contrast motivational measures have been a central aspect of behavioural research in social psychology. These provide quantitative scales of public opinion relating to a specified behaviour and the basis for agreement or disagreement with a behaviour. Where the aim is to seek affirmation of a management option affecting public behaviour then social psychology offers more insight than economic valuation. For example, public support for policy initiatives, such as recycling or car sharing, can be explained directly on this basis. Models in social psychology separate general and specific attitudes, social norms, and behavioural action measures, e.g., perceived behavioural control (Fishbein and Ajzen, 1975; Ajzen, 1991). These models are widely applied in other areas.

A more neglected aspect is that of ethical norms which can also be categorisation for use in analysing behaviour e.g. as applied to understanding intended WTP (Spash, 2000a,b). The overall result is to broaden the model of environmental valuation well beyond that commonly considered in economics. For example, that individuals state a WTP to conserve species they will never see is more credibly understood as an expression of ethical beliefs, such as rights, which do not require all well-being to be expressed via the purchase of an hedonic experience (Kahneman and Sugden, 2005 note this limitation with respect to experience utility measures). Investigating such models leads to an acceptance that individuals hold multiple values when considering environmental entities and quality change (Spash, 2000c). The problem then becomes how to design institutional processes which allow different values to be expressed (Vatn, 2005).

Brouwer (2000) has advocated a specific set of value transfer steps which are in effect an institutional process. His seven stages are: (i) define the environmental goods and services; (ii) identify stakeholders; (iii) identify values held by different stakeholder groups; (iv) involve stakeholders in the determination of monetary environmental value validity; (v) select primary study data taking into account internal and external validity; (vi) account for primary study design impacts on value outcomes; and (vii) involve stakeholders in the validation of values being transferred. The conclusion to the process is then noted as follows (Brouwer, 2000: 150): “Finally, the economic aggregate is included in a CBA together with other economic costs and benefits, which can then play its part in the facilitation of the overall, real world, multi-criteria decision-making process”. Thus, both participation and MCA are recognised by

Brouwer as necessary parts of the overall decision process and value transfer is placed within that context.

MCA covers the range of methods developed to explicitly include multiple values. The basic method requires setting out project or policy options, determining specific performance criteria, and evaluating each option relative to the criteria. Criteria, such as distributional impacts, economic returns, environmental damage and so on, may be weighted by importance. Criteria can be treated as incommensurable. Different MCA approaches vary in their weighting, summing and aggregating procedures and their theoretical basis (for a review see De Montis et al., 2004). MCA can be used to explain reasons for conflict between stakeholders (Munda, 1995) and to unravel such things as judgements on genetically modified crops (Stirling and Mayer, 2001). MCA approaches can be compatible with monetary valuation or value transfer as these can be criteria in the decision matrices. Indeed the outcome of good sensitivity analysis under CBA is in effect a type of MCA (see Merrifield, 1997). The 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, or in Brouwer's case "the real world, multi-criteria decision-making process".

The use of open MCA which address conflicts has led to such methods being combined with participatory approaches. Stakeholder or vested interest groups can be brought together in different formats and results analysed to understand why conflicts arise and to aid consensus seeking. Methods such as mediated modelling, scenario analysis, and social multi-criteria evaluation have been used in this way (Kallis et al., 2006).

Interest in "participatory approaches" has been spreading,[7] and in the environmental policy arena, and elsewhere, there has been a push for greater public participation, (e.g., the Aarhus Convention, European Commission, 1998) and the inclusion of non-governmental stakeholders in project appraisal (Beierle and Konisky, 2001). Focus groups, citizens juries and consensus conferences are all methods used to aid decision processes using deliberation in small groups (De Marchi and Ravetz, 2001). Of course they also have their own problems (Spash et al., 2004), such as whether attendees represent individuals, social groups or organisations (O'Neill, 2001). The point here is that there are real alternatives which may often prove more defensible, and are not necessarily more expensive, than some of the current work producing numbers from value transfer approaches.

Participatory approaches have also been regarded as useful for economic analysis in a range of different ways best summarised as "deliberative monetary valuation" (Spash, 2001). These may either supplement or replace the CVM. Macmillan et al. (2002) use small group discussion before administering a CVM survey on an individual basis and claim standard WTP is improved. In contrast, Jacobs (1996) has advocated deliberation for obtaining the willingness of a group to have society pay. Norton (1998) sees community based discussions over ecosystems management of long term impact as distinct from CBA and recommends a non-welfare multi-scalar index to complement standard approaches. Kaplowitz and Hoehn (2001) show how focus groups provide complementary information on how individuals value the environment compared to individual interviews. There is then a concern that conducting deliberation followed by reducing the information to an individual WTP would exclude a whole range of information.

Indeed, community and group based approaches move well outside the standard individualistic foundation of economic theory. There are several fundamental differences between political science and economic approaches and concerns have been raised over attempts to justify the latter by using elements of the former (Niemeyer and Spash, 2001). Group deliberation has been argued to lead to a revelation of more fundamental values and evidence shows individuals can go beyond their private self interest positions with deliberation being a transformation process (Niemeyer, 2004). The ideal from the political science perspective appears to be the design of value articulating institutions to promote social equity (Dryzek, 1990), rather than the production of individual's WTP.

In summary, there is no one best approach, but rather an interesting array of alternatives for addressing policy problems. For example, consider designing policy instruments for nitrate non-point pollution control in water bodies. No cause–effect relationship exists between a farmers production system and the impacts of nitrates in the water body. Value transfer would recommend transferring an arbitrary cause–effect model and transferring uncertain monetary benefits of nitrate reduction to attempt estimating an economically efficient nitrate level. Alternatively we might assume an arbitrary farm nitrate reduction and focus research on the impacts of different instruments on farmers' behaviour using approaches form social psychology to attempt an effective policy design. However, contentious issues and policy conflicts would be best addressed explicitly. Scientific uncertainties might be approached using mediated modelling to achieve stakeholder engagement and improved understanding or management strategies might be explored via a MCA. For those concerned about improving environmental decision processes, monetary value transfers are but one, often very imperfect, approach which closes down environmental problems when they may require opening-up.

6. Conclusions

Information transfer in CBA will typically involve both natural science data, say on cause–effect relationships, and economic data, say on costs of capital. Benefit transfer is a sub-category of value transfer which in turn is a sub-category of information transfer. The contentious nature of different types of information transfer relates to the extent to which they can be validated using standard scientific procedures. As explained here, the move towards aggregated, systems level value transfer (i.e. ecosystems service valuation) moves well beyond normal science and controlled repeated experimental validity testing. Both value transfer and primary studies also have a serious range of caveats which must be taken into account. Overall results from convergent validity tests show that the uncertainty

[7] Research on deliberation and participation in the environmental area has been ongoing for a good 20 years and some of the leaders in this respect are Jacquie Burgess, Tim O'Riordan, Ortwin Renn and John Dryzek.

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. A sound knowledge is required as to the intended use of values, the expected level of accuracy and robustness required, and their comparability to alternatives which might achieve the same ends.

While reviewing a substantial part of the value transfer literature we became concerned but also hopeful. The worry relates to the fact that including socio-economic, demographic and attitudinal variables in transfer functions may be at odds with the very theoretical foundation of the valuation studies involved, and that this problem seems largely unobserved. The optimism relates to the idea that, by learning about the challenges involved when taking on board such variables, a richer theory of preference formation and choice may actually be identified and evolve — a theory which recognises that preferences are to a substantial degree socially and culturally shaped. We believe the inclusion of a wide range of socio-economic variables is reasonable given the need to understand variations in the appraisals of different environmental goods. At the same time, this does clearly go beyond the bounds of methodological individualism.

The use of value transfer needs to be more carefully considered in terms of both what is desired by decision processes and what alternatives can offer. Alternatives do need to show their ability to address the problems faced by those in the policy process who demand value transfer studies, and should also be able to avoid the theoretical issues raised by value transfer studies. The general point is that the context in which values are intended to be used determines their acceptability. There are now a serious range of alternatives available for assessing environmental values, concerns and conflicts. This is not to deny that these also have their own caveats but rather to note the need for serious consideration as to the best method for any given issue, policy context or problem.

Environmental values as estimated by monetary valuation are one specific class of values and they need to be seen as such. Economists do often recognise this in passing but rarely make attempts to be more explicit, which they must do if they are to act as policy advisors. Some ecologists have jumped on the monetary valuation band wagon without recognising the limited scope of and validity attributable to the economic value estimates they then transfer regardless of content or meaning. Clearly valuation is an interdisciplinary undertaking linking natural science with social science, and as such a full range of perspectives on human behaviour is required including social psychology, political science, sociology, and applied philosophy. Improved understanding of environmental values is needed along with institutions which are capable of supporting people in expressing their values in ways they find to be sound. Where policy makers demand theoretically meaningless numbers, on grounds of pragmatism, they need to be challenged rather than pandered to.

REFERENCES

Ajzen, I., 1991. The theory of planned behaviour. Organizational Behavior and Human Decision Processes 50, 179–211.

Aldred, J., 2002. Cost–benefit analysis, incommensurability and rough equality. Environmental Values 11, 27–47.

Barton, D.N., 2002. The transferability of benefit transfer: contingent valuation of water quality improvements in Costa Rica. Ecological Economics 42, 147–164.

Bateman, I., Brainard, J., Jones, A., Lovett, A., 2005. Geographical Information Systems (GIS) as the last/best hope for benefit function transfer. Benefit Transfer and Valuation Databases: Are We Heading in the Right Direction. United States Environmental Protection Agency and Environment Canada, Washington, D.C.

Batker, D., Barclay, E., Boumans, R., Hathaway, T., Burgess, E., Shaw, D., Liu, S., 2005. Ecosystem Services Enhanced by Salmon Habitat Conservation in the Green/Duwamish and Central Puget Sound Watershed. Asia Pacific Environmental Exchange, Tacoma, p. 92.

Becker, G.S., 1976. The Economic Approach to Human Behavior. University of Chicago Press, Chicago.

Beierle, T.C., Konisky, D.M., 2001. What are we gaining from stakeholder involvement. Observations from environmental planning in the Great Lakes? Environment and Planning. C, Government and Policy 19 (4), 515–527.

Bockstael, N.E., Freeman, A.M., Kopp, R.J., Portney, P.R., Smith, V.K., 2000. On measuring economic values for Nature. Environmental Science and Technology 34, 1384–1389 (April).

Brainard, J., Lovett, A., Bateman, I.J., 1999. Integrating geographical information systems into travel cost analysis and benefit transfer. International Journal of Information Science 13 (3), 227–246.

Brouwer, R., 2000. Environmental value transfer: state of the art and future prospects. Ecological Economics 32, 137–152.

Chang, R. (Ed.), 1997. Incommensurability, Incomparability and Practical Reason. Harvard University Press, Harvard.

Chattopadhyay, S., 2003. A repeated sampling technique in assessing the validity of benefit transfer in valuing non-market goods. Land Economics 79 (4), 576–596.

de Groot, R., Wilson, M., Boumans, R., 2002. A typology for the classification, description, and valuation of ecosystems goods and services. Ecological Economics 41, 393–408.

De Marchi, B., Ravetz, J.R., 2001. Participatory approaches to environmental policy. In: Spash, C.L., Carter, C. (Eds.), EVE Policy Brief. Cambridge Research for the Environment, Cambridge, p. 18.

De Montis, A., De Toro, P., Droste, B., Omann, I., Stagl, S., 2004. Assessing the quality of different MCDA methods. In: Getzner, M., Spash, C.L., Stagl, S. (Eds.), Alternatives for Environmental Valuation. Routledge, London.

Diamond, P.A., Hausman, J.H., 1994. Contingent valuation: is some number better than no number? Journal of Economic Perspectives 8 (4), 45–64.

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

Erickson, J.D., 1993. From ecology to economics: the case against CO_2 fertilization. Ecological Economics 8 (2), 157–176.

European Commission, 1998. Aarhus Convention on Access to Information, Public Participation in Decision Making and Access to Justice in Environmental Matters. Brussels.

Fishbein, M., Ajzen, I., 1975. Belief, Attitude, Intention and Behavior: An Introduction to Theory and Research. Addison-Wesley, Reading, Massachusetts.

Funtowicz, S.O., Ravetz, J.R., 1993. Science for the post-normal age. Futures 25 (7), 739–755.

Funtowicz, S.O., Ravetz, J.R., 1994. Uncertainty, complexity and post-normal science. Environmental Toxicology and Chemistry 13 (12), 1881–1885.

Green, C., 2004. Benefit Transfer. Middlesex University, London, p. 38.

Hausman, J.A. (Ed.), 1993. Contingent Valuation: A Critical Assessment. North-Holland, Amsterdam.

Jacobs, M., 1996. Environmental valuation, deliberative democracy and public decision-making institutions. In: Foster, J. (Ed.), Valuing Nature? Ethics, Economics and Environment, vol. 13. Routledge, London, pp. 211–231.

Jiang, Y., Swallow, S.K., Mcgonagle, M.P., 2005. Context sensitive benefit transfer using stated choice models: specification and convergent validity for policy analysis. Environmental and Resource Economics 31, 477–499.
Kahneman, D., Sugden, R., 2005. Experienced utility as a standard of policy evaluation. Environmental and Resource Economics 32 (1), 161–181.
Kallis, G., Videira, N., Antunes, P., Guimarães Pereira, Â., Spash, C.L., Coccossis, H., Corral Quintana, S., del Moral, L., Hatzilacou, D., Lobo, G., Mexa, A., Paneque, P., Pedregal, B., Santos, R., 2006. Participatory methods for water resource planning. Environment and Planning. C, Government and Policy 24 (2), 215–234.
Kaplowitz, M.D., Hoehn, J.P., 2001. Do focus groups and individual interviews reveal the same information for natural resource valuation. Ecological Economics 36 (2), 237–247.
Knetsch, J.L., 1994. Environmental valuation: some problems of wrong questions and misleading answers. Environmental Values 3 (4), 351–368.
Knetsch, J.L., 2005. Gains, losses, and the US EPA economic analyses guidelines: a hazardous product. Environmental and Resource Economics 32 (1), 91–112.
Kotchen, M.J., Reiling, S.D., 2000. Environmental attitudes, motivations, and contingent valuation of nonuse values: a case study involving endangered species. Ecological Economics 32 (1), 93–107.
Loomis, J.B., 1992. The evolution of a more rigorous approach to benefit transfer: benefit function transfer. Water Resources Research 28 (3), 701–705.
Macmillan, D.C., Philip, L., Hanley, N., Alvarez-Farizo, B., 2002. Valuing the non-market benefits of wild goose conservation: a comparison of interview and group-based approaches. Ecological Economics 43 (1), 49–59.
Martinez-Alier, J., Munda, G., O'Neill, J., 1998. Weak comparability of values as a foundation for ecological economics. Ecological Economics 26 (3), 277–286.
McFadden, D., Leonard, G.K., 1993. Issues in the Contingent Valuation of Environmental Goods: Methodologies for Data Collection and Analysis. In: Hausman, J.H. (Ed.), Contingent Valuation. A Critical Assessment. North Holland, Amsterdam, pp. 165–208.
Merrifield, J., 1997. Sensitivity analysis in benefit–cost analysis: a key to increased use and acceptance. Contemporary Economic Policy XV, 82–92 (July).
Morrison, M., Bennett, J., Blamey, R., Louviere, J., 2002. Choice modelling and tests of benefit transfer. American Journal of Agricultural Economics 84 (1), 161–170.
Munda, G., 1995. Multicriteria Evaluation in a Fuzzy Environment. Theory and Applications in Ecological Economics. Physica-Verlag, Heidelberg.
Navrud, S., Bergland, O., 2001. Value transfer and environmental policy. In: Spash, C.L., Carter, C.C. (Eds.), Environmental Valuation in Europe, vol. 8. Cambridge Research for the Environment, Cambridge, p. 18.
Niemeyer, S., 2004. Preference transformation through deliberation: protecting world heritage. In: Getzner, M., Spash, C.L., Stagl, S. (Eds.), Alternatives for Environmental Valuation. Routledge, London, pp. 263–289.
Niemeyer, S., 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.
Norton, B.G., 1998. Evaluation and ecosystem management: new directions needed? Landscape and Urban Planning 40 (1–3), 185–194.
O'Neill, J., 1997. Value, Pluralism, Incommensurability and Institutions. In: Foster, J. (Ed.), Environmental Economics: A Critique of Orthodox Policy. Routledge, London.
O'Neill, J., 2001. Representation. Environment and Planning. C, Government and Policy 9 (4).
Price, C., 1993. Time, Discounting and Value. Basil Blackwell, Oxford, England.
Ready, R., Navrud, S., Day, B., Dubourg, R., Machado, F., Mourato, S., Spanninks, F., Rodriquez, M.X.V., 2004. Benefit transfer in Europe: how reliable are transfers between countries? Environmental and Resource Economics 29, 67–82.
Rosenberger, R.S., 2005. Publication measurement error in benefit transfers. Benefit Transfer and Valuation Databases: Are We Heading in the Right Direction. United States Environmental Protection Agency and Environment Canada, Washington, D.C.
Rozan, A., 2004. Benefit transfer: a comparison of WTP for air quality between France and Germany. Environmental and Resource Economics 29, 295–306.
Shrestha, R.K., Loomis, L.B., 2003. Meta-analytic benefit transfer of outdoor recreation economic values: testing out-of-sample convergent validity. Environmental and Resource Economics 25, 79–100.
Smith, V.K., Van Houten, G., Pattanayak, S.K., 2002. Benefit transfer via preference calibration: "Prudential algebra" for policy. Land Economics 78 (1), 132–152.
Spash, C.L., 1997a. Assessing the economic benefits to agriculture from air pollution control. Journal of Economic Surveys 11 (1), 47–70.
Spash, C.L., 1997b. Reconciling different approaches to environmental management. International Journal of Environment and Pollution 7 (4), 497–511.
Spash, C.L., 2000a. Ecosystems, contingent valuation and ethics: the case of wetlands re-creation. Ecological Economics 34 (2), 195–215.
Spash, C.L., 2000b. Ethical motives and charitable contributions in contingent valuation: empirical evidence from social psychology and economics. Environmental Values 9 (4), 453–479.
Spash, C.L., 2000c. Multiple value expression in contingent valuation: economics and ethics. Environmental Science and Technology 34 (8), 1433–1438.
Spash, C.L., 2001. Deliberative Monetary Valuation. 5th Nordic Environmental Research Conference. University of Aarhus, Denmark.
Spash, C.L., 2002. Greenhouse Economics: Value and Ethics. Routledge, London.
Spash, C. L., in press. Non-economic motivation for contingent values: Rights and attitudinal beliefs in the willingness to pay for environmental improvements. Land Economics 82 (4).
Spash, C.L., McNally, S., 2001. Managing Pollution: Economic Valuation and Environmental Toxicology. Edward Elgar, Cheltenham.
Spash, C.L., O'Neill, J., 2000. Conceptions of value in environmental decision-making. Environmental Values 9 (4), 521–536.
Spash, C.L., Stagl, S., Getzner, M., 2004. Exploring alternatives for environmental valuation. Alternatives for Environmental Valuation. In: Getzner, M., Spash, C.L., Stagl, S. (Eds.), Routledge, London.
Stigler, G.J., Becker, G.S., 1977. De gustibus non est disputandum. American Economic Review 76 (1), 76–90.
Stirling, A., Mayer, S., 2001. A novel approach to the appraisal of technological risk: a multi-criteria mapping study of a genetically modified crop. Environment and Planning C: Government and Policy 19 (4), 529–555.
Svedsater, H., 2003. Economic valuation of the environment: how citizens make sense of contingent valuation questions. Land Economics 79 (1), 122–135.
Vatn, A., 2004. Environmental valuation and rationality. Land Economics 80 (1), 1–18.
Vatn, A., 2005. Institutions and the Environment. Edward Elgar, Cheltenham.
Vatn, A., Bromley, D.W., 1994. Choices without prices without apologies. Journal of Environmental Economics and Management 26 (2), 129–148.