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Ecosystem valuation

A sequential decision support system and quality assessment issues

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Understanding the economic value of nature and the services it provides to humanity has become increasingly important for local, national, and global policy and decision making. It has become obvious that quantifying and integrating these services into decision making will be crucial for sustainable development. Problems arise in that it is difficult to obtain meaningful values for the goods and services that ecosystems provide and for which there is no formal market. A wide range of ecosystem services fall into this category. Additional problems arise when economic methods are applied inappropriately and when the importance of ecosystem maintenance for human welfare is underestimated. In this article we identify a place for monetary valuation within the pluralistic approach supported by ecological economics and assess progress to date in the application of environmental valuation to ecosystem service provision. We first review definitions of ecosystem services in order to make an operational link to valuation methods. We then discuss the spatially explicit nature of ecosystem services provision and benefits capture. We highlight the importance of valuing marginal changes and the role for macroscale valuation, nonlinearities in service benefits, and the significance of nonconvexities (threshold effects). We also review guidance on valuation studies quality assurance, and discuss the problems inherent in the methodology as exposed by the findings of behavioral economics, as well as with *benefits transfer*—the most common way valuation studies are applied in the policy process. We argue for a sequential decision support system that can lead to a more integrated and rigorous approach to environmental valuation and biophysical measurement of ecosystem services. This system itself then needs to be encompassed within a more comprehensive multicriteria assessment dialogue and process.

Keywords: ecosystem services; ecosystem valuation; spatial explicitness; double counting; marginality; threshold effects; nonlinearities; ecosystem valuation techniques

Introduction

A core principle of ecological economics is methodological pluralism[6] and the analysis presented below takes that as axiomatic. Although the focus of attention is on monetary valuation methods and techniques within an extended cost-benefit framework, this should not be interpreted as a plea for dominance over other evaluation approaches. Rather the objective is to review progress to date, recognize limitations, and indicate future prospects for the monetary valuation methodology. The line of argument pursued is also not one in which economic efficiency is given meta-ethical standing, but neither is it dismissed as irrelevant as environmental goods and services become increasingly scarce. Cost-benefit analysis suitably adjusted for equity concerns can, we would argue, still play an important (though often not decisive) role in multicriteria assessment decision support systems.[6,7] Both the full commodification of the environment and the assignment of monetary values to all aspects of its complex functioning and existence is not a sound scientific or moral basis for sustainable environmental management and policy. Nevertheless, in real policy contexts it is the case that tradeoffs are continually made between conservation and development options and monetary valuation and opportunity cost calculations can and do play a useful heuristic role within a set of constraints.[8,9] The constraints set should encompass, among others, scientific uncertainties surrounding complex ecosystems and tipping point dangers and ethical concerns over human rights, interests and livelihoods (especially of the poor), and nonhuman interests.

Despite increasing recognition of the importance of ecosystems and biodiversity for human welfare, they continue to decline at an unprecedented rate.[1,10–13] In many cases the losses are irreversible, posing a serious threat to sustainable development and to human well-being in general.[1,14] In view of this, ecologists and economists have made considerable effort to increase the understanding of the importance of ecosystems in order that this is better reflected in decisions which affect their maintenance and services provision. Despite a dramatic increase in the number of studies aiming to value ecosystem services, there appears to be "growing confusion amongst decision-makers, ecologists and noneconomists about the validity and implications of ecosystem valuation" (p. 2).[15]

The valuation of ecosystem services is a complex process that is reliant on the availability of relevant and accurate biophysical data on ecosystem processes and functions, but also on the correct and appropriate application of economic valuation techniques, alongside other valuation methods. Not all studies meet these requirements. This presents a challenge to policy makers looking to use the results of valuation studies but who may be less well equipped to distinguish between studies providing reasonably robust results, and those which yield crude, and in some instances, unsound value estimates. In particular, this paper aims to provide clarification to practitioners on: (i) important considerations in ecosystem services identification and valuation; (ii) how the ecosystem valuation literature has dealt with these issues to date and the limitations of the approach as exposed by behavioral economics findings; (iii) guidance on how to assess the internal quality of valuation studies; and (iv) on the appropriate use of benefits transfer. We focus first on ecosystem services definitions.

Ecosystem services

Many definitions and classification schemes for ecosystem services exist.[1,5,16,17] One of the most widely cited is the Millennium Ecosystem Assessment definition, which describes ecosystem services as "the benefits that people obtain from ecosystems." It classifies services into *supporting, regulating, provisioning,* and *cultural*. This framework provides an excellent platform for moving toward a more operational classification system, which explicitly links changes in ecosystem services to changes in human welfare. By adapting and reorienting this definition it can be better suited to the purpose at hand, with little loss of functionality. Wallace,[18] for example, has focused on land management, while Boyd and Banzhaf[19] and Maler *et al.*[20] take national income accounting as their policy context. Boyd and Banzhaf[19] use the definition that "final ecosystem services are components of nature, directly enjoyed, consumed or used to yield human well-being." Critically, the authors highlight that "services" and "benefits" are different, a "benefit" being something that may be based on ecosystem service inputs, but is not an ecosystem service itself. Moreover, services are considered to be ecological things or characteristics, rather than functions or processes (p. 620).[19] Building on this, Fisher and Turner[21] proposed that "ecosystem services are the aspects of ecosystems utilized (actively or passively) to produce human well-being" (p. 1167). Their definition of services differs from that of Boyd and Banzhaf[19] in that they consider processes or functions to be services as long as there are human beneficiaries. Also an intermediate service is one that influences human well-being indirectly, whereas a final service contributes directly. Classification is context dependent—for example, clean water provision is a final service to a person requiring drinking water, but it is an intermediate service to a recreational angler. Importantly, a final service is often but not always the same as a benefit. For example, recreation is a benefit to the recreational angler, but the final ecosystem service is the provision of the fish population. This approach seeks to provide a transparent method for identifying the aspects of ecosystem services which are of direct relevance to economic valuation, and critically, to avoid the problem of double-counting. Moreover, it was developed in order to meet the requirements of a real case study that is investigating the value of ecosystem services provided by the Eastern Arc Mountains, Tanzania.[22]

Important considerations in economic valuation

We now consider a number of issues highlighted by economists as critical to the *appropriate* economic valuation of ecosystem services, namely: spatial explicitness, marginality and macrovalues, the double-counting trap, nonlinearities in benefits, and threshold effects.

It is critically important to first and foremost clarify the level of understanding (or ignorance) of underlying biophysical structure and processes through spatially explicit models of any given ecosystem service. This contextual analysis must then encompass appropriate socio-economic, political, and cultural parameters in order to properly identify ecosystem services supply and demand side beneficiaries. This scoping exercise also serves to highlight the "political economy" context and the need to recognize a plurality of values that may not be strictly commensurable.

At the operational level the valuation of ecosystem services is often required "at the margin."[a] This means focusing on relatively small, incremental changes rather than large state changing impacts.[3,4,23–25] To illustrate why this is important we use Figure 1 which shows the marginal supply (cost) and marginal benefit (demand) curves for an essential ecosystem service.[17] The vertical axis measures economic value or price, and the horizontal axis measures the flow or quantity of ecosystem service. The marginal supply curve is vertical because the supply of ecosystem services does not tend to increase or decrease in relation to economic systems.[17] The downward-sloping demand curve shows how much individuals are willing to pay for an incremental amount of ecosystem service. The shape indicates that the more scarce the ecosystem service, the more an additional unit is valued (or, conversely, the more of a service we have, the less we value an additional unit). Given ecosystem services are only

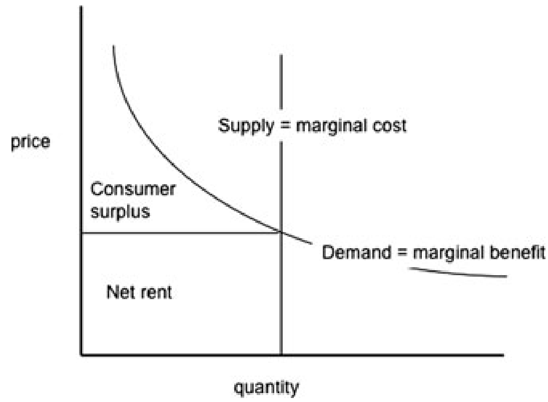

Figure 1. **Stylized supply and demand curves for ecosystem services.**

substitutable up to a point, as the quantity available approaches zero (or some necessary minimum provision of ecosystem structure and processes) the demand curve approaches infinity. The area beneath the demand curve indicates the total economic value of the ecosystem service. However, as the curve is not bounded on the horizontal axis, this area can not be fully defined. Policy decisions related to natural resources often involve tradeoffs, which occur at the margin. This means examining the value of the "next unit" at some point along the marginal benefit curve.[24,26] However, given the scientific uncertainties that shroud ecosystem functioning, it is often difficult to discern whether a given change is "marginal" or not and when thresholds are being approached or crossed.[4] At the strategic level, a more macroscale valuation may play an indicative role in decision making.[27] It can contribute to the further development of indicators of human welfare and sustainability.[28] The objective would be to estimate the expenditure that would be required to purchase available ecosystem services at their shadow prices. Howarth and Farber[27] argue that this approach is analogous to the concept of gross domestic product, which is the total value of market goods and services evaluated via market prices.

Given ecosystem services cross scales, when deciding whether the "next unit" is meaningful in terms of marginal analysis it becomes important to consider the scale of the policy decision.[29] For example, at the local scale, the loss of an entire forest on which livelihoods are dependent, may be so catastrophic as to render the change meaningless

[a]Specifically, the focus on small-scale changes was to avoid large income effects (which may change the marginal utility of income) and violations of the *ceteris paribus* assumption (changes in other prices).[97]

(since lives would be so dramatically altered). A more relevant change may be to consider the loss of *part* of the forest. Conversely, at the global scale, the loss of an entire forest may be "marginal"—at this scale it is not likely to be perceived as a catastrophic loss for life support or international economic systems. In practice, applying the concept to multiscale systems may be confusing. As a guide, Fisher *et al.*[29] recommend considering the "next unit" in terms of the geographic extent a policy decision could encompass, for example, extending a forest within its national borders.

Another widely recognized issue concerns the potential problem of double-counting.[1,4,19,24,30,31] This may occur where competing ecosystem services are valued separately and the values aggregated; or, where an intermediate service is first valued separately, but also subsequently through its contribution to a final service benefit. For example, the value of a forest ecosystem for clearance timber logging should not be added to the value of the same forest patch for recreational benefits since the former will likely preclude the latter. Nor should the value of a pollination service, which is already embodied in the market price of a crop, be counted separately unless the value of its input to the crop is deducted. In essence, double-counting is a feature of the complexity of ecosystem services and the difficulty in understanding their multiple interactions. Unfortunately, there are numerous cases where researchers have incorrectly summed values in order to obtain aggregate estimates of ecosystem value (evidence from Fisher *et al.*[29]). It is thus essential that the analyst has a clear understanding of the various overlaps and feedbacks between services when undertaking aggregation.[24,31] Hein *et al.*[32] suggest only including regulation services in valuations if "(i) they have an impact outside the ecosystem to be valued; and/or (ii) if they provide a direct benefit to people living in the area (i.e., not through sustaining or improving another service)" (p. 214). Alternatively, the classification scheme recommended by Fisher and Turner[21] helps to avoid the problem by drawing a clear distinction between intermediate services, final services, and benefits, the latter being the focus of economic valuation.

The existence of nonlinearities in ecosystem services provision adds further complexity to their valuation and subsequent management. Because many ecosystems typically respond nonlinearly to disturbances, their supply may seem to be relatively unaffected by increasing perturbation, until they suddenly reach a point at which a dramatic system-changing response occurs—for example, in the ecology of phosphorus-limited shallow lakes, which can flip suddenly from one state to another.[24,33] Further, in situations where nonlinearities occur, one can not make the assumption that marginal benefit values are equally distributed. For example, the storm protection benefit of a unit increase in mangrove habitat area may not be assumed to be constant for mangroves of all sizes due to nonlinearities in wave attenuation.[34] If a cost-benefit appraisal assumes linearity, but service provision is in fact nonlinear, policy option outcomes may be unnecessarily polarized. Correspondingly, for ecosystem valuation to better inform policy decisions, nonlinearities need to be clearly understood and reflected in both ecological and economic analysis. Threshold effects pose especially complex policy and analysis challenges.

The threshold effect refers to the point at which an ecosystem may change abruptly into an alternative steady state.[35,36] For marginal analysis to hold true, the "next unit" to be valued should not be capable of tipping the system over a functional threshold or "safe minimum standard" (SMS).[4,29,37] From an ecosystem valuation perspective, the SMS represents the minimum level of a well-functioning ecosystem that is capable of producing a sustainable supply of service. Conceptually, looking at Figure 1, this means that marginal analysis should only be conducted far away from the point at which the demand curve increases sharply to infinity. In practice, this requires knowledge of the location of the SMS zone. Of course, due to the considerable uncertainty surrounding ecosystem function, this introduces complexity since it is often far from clear when a threshold may be reached.[24,38] Identifying this hazardous zone will require expert input from ecologists, risk analysts, and others.

In situations of uncertainty and near thresholds, marginal analysis will not be appropriate; instead, more precautionary approaches will be required, such as the SMS approach. This goes back to the earlier work of Ciriacy-Wantrup[39] and, later, Bishop,[40] who put forth the argument that a project should be rejected if irreversible losses of nature could occur, unless the social costs of doing so were prohibitive. In other words, putting conservation first, because

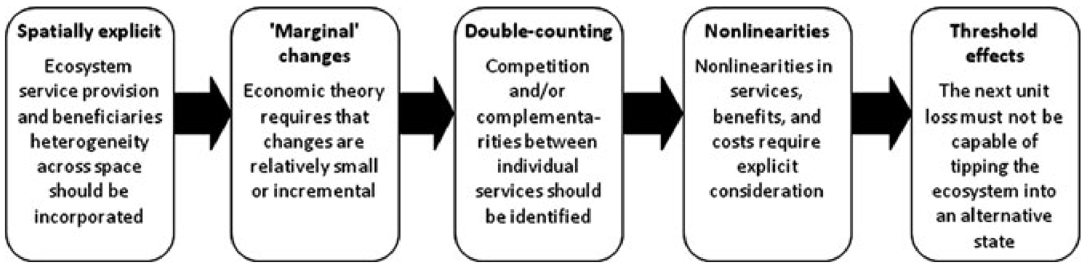

Figure 2. Sequential decision support system.

of functional transparency,[38] needs to be balanced against the costs of doing so. This cost-based assessment can be informed by a range of data and values.

Ultimately, in the face of uncertainty and potential irreversibility, decisions regarding ecosystems will require ethical and political choices to be made and deliberately agreed. In the longer term, the solution would be to improve our understanding of ecosystem function so we can move toward more situations where we are dealing with risk rather than uncertainty. In the meantime, recent work by Lenton *et al.*[41] proposes the use of early warning systems which identify systems that are likely to cross "tipping points" (threshold effects) and are relevant to policy and accessed by humans ("tipping elements"), historical data and predictive modeling (e.g., degenerative fingerprinting) may then be used to locate tipping points.

In summary, to be most useful for policy, services must be assessed within their appropriate spatial context and economic valuation should provide estimates of value (avoiding double counting) that can feed into decisions at the appropriate scale, and which recognize possible nonlinearities and are well within the bounds of SMS. Figure 2 summarizes these sequential steps that we argue are the necessary and sufficient elements in any ecosystem services assessment that is a component within a more comprehensive decision support system. We now review selected examples to highlight why these issues are important and how the ecosystem valuation literature has dealt with them to date (see Table 1).

Ecosystem services valuation in practice

Spatial explicitness

The requirement for spatially explicit ecosystem valuation is based on recognition that ecosystem services are context dependent in terms of their provision and their associated benefits and costs. The importance of this point is neatly illustrated by Naidoo and Ricketts[42] in a cost-benefit analysis of three potential equivalent conservation corridors in Mbaracayu Biosphere Reserve, Eastern Paraguay. One corridor generated net benefits three times greater than the other corridors. The disparity was largely due to differences in opportunity costs as a result of variability in spatial factors, such as land tenure, slope, and soil type. For example, protected areas, indigenous reserves, and areas with steeper slopes generated significantly lower opportunity costs due to lower conversion rates. The benefits generated by two of the five[b] ecosystem services were also found to differ significantly between corridors due to spatial variables. For example, bushmeat benefits varied from \$0/ha to \$18.50/ha mainly due to forest patch size, while carbon storage values varied according to forest type. Explicitly incorporating the spatial context into the cost-benefit analysis was critical in obtaining estimates of both the costs and benefits of ecosystem provision, and, crucially, in enabling conservation planners to identify the most economically efficient location for the conservation corridors.

In a different example, Luisetti *et al.*[43] illustrate the importance of spatial context in aggregating benefits of new wetland creation on the east coast of England. The authors used a site-specific choice experiment survey to elicit ecosystem service values from the regional population closest to the proposed scheme. The good was described in terms of five attributes: salt marsh area, number of bird species observable, distance from home, accessibility and

[b]Data limitations prevented the spatial analysis of timber harvesting and bioprospecting services, while existence values were deemed to be spatially homogenous at the scale of analysis.

Table 1. **Ecosystem services valuation techniques, limitations, quality indicators, and selected examples**

Technique	Approach	Limitations	Quality Checks/Indicators	Selected Examples
Market Prices	A simple accounting procedure to value environmental goods or services which are traded in *markets* (can also be extended to other *nonmarket* ecosystem service benefits by observing how changes in provision affect the prices or quantities of other marketed goods)	• Only applicable where market data available • Market price may not offer true reflection of marginal social costs and benefits • Lower bound estimates • Sensitive to functional form	• Any price distortions due to market imperfections or policy failure should be corrected • Assessment of market capacity included • Examination of changes in real prices over time • Appropriate functional form for demand curve	Nontimber forest products and timber goods;[46,50] fish nursery services of proposed managed realignment scheme;[43] river in stream flows for agricultural supply;[111] wetland productivity function for commercial fishing[112]
Production Function (also known as *dose–response* technique)	Involves tracing the impact of a physical change in the quantity or quality of an ecosystem service along a series of pathways to ascertain the corresponding impact on human welfare[113]	• Data is often lacking on change in service and consequent impact on production • Can not estimate nonuse values	• Utilization of expert scientific knowledge of ecosystem functions • Explicit cause and effect modeling (not just correlation) incorporating possible threshold levels and discontinuities • Modeling of whole market (demand and supply) including dynamic effects • Prices of all inputs and outputs corrected for distortions • Absence of double-counting in studies on multiple use systems[114]	Forest-based pollination services for coffee production;[115] wetland groundwater recharge for irrigated agriculture;[116] nursery and breeding habitat function of coastal wetlands;[113] forest watershed protection services for groundwater recharge[117]

Continued.

Table 1. *Continued*

Technique	Approach	Limitations	Quality Checks/Indicators	Selected Examples
Travel Cost Method (TCM)	Survey based technique using information on observed travel and time expenditures (a central assumption is that the benefit an individual receives from a particular site is worth at least as much as he or she is willing to pay to visit it)	• Applicable only in a few contexts • Requires large amount of data • Complex when trips are multipurpose • Can not estimate nonuse values	• Reasonable site definition, spatially explicit and coverage of entire area to be affected[91] • Modeling of participation: inclusion of nonvisitors as well as visitors • Site selection which reflects actual choice sets • Inclusion of site-specific data on services, lodging options and communication • Exclusion of indirect costs from travel cost variables and cost of equipment used one more than one occasion[91] • Appropriate estimation of shadow price for time • Appropriate and relevant selection of environmental quality variable, ideally in quantitative terms. • Consideration of and appropriate adjustment for multipurpose trips • Model explanatory power and confidence intervals for environmental quality attribute and travel cost • More robust results may be achieved in studies which combine TCM and CVM or choice modeling	Recreational benefits from greater water deliveries to wetlands;[118] recreational benefits from hypothetical release of dams;[119] recreational fishing in Brazilian Pantanal[120] and in Stockholm Archipelago[58]

Continued.

Table 1. *Continued*

Technique	Approach	Limitations	Quality Checks/Indicators	Selected Examples
Hedonic Pricing (HP)	Assumes the good of interest may be *implicitly traded via demand for a marketed good*; in most cases this will be in the property market, e.g., scenic beauty is often implicitly traded such that its value may be calculated by the price differential between two identical houses where one is located in an area of outstanding natural beauty and the other is not[121]	• Dependent on large amount of data • Very sensitive to specification • Can not estimate nonuse values	• Price data based on individual transactions in market rather than assessed values[91] • Consideration of measurement error in price data and appropriate adjustment • Correct specification of HP function and availability of accurate data for all variables[91,122] • Appropriate and relevant selection of environmental quality variable, ideally in quantitative terms • Appropriate functional form: linear models typically inadequate • Checks for multicollinearity and appropriate action[106,122] • Correct definition of market extent—under- rather than overestimated[122,123] • Model explanatory power and confidence intervals for environmental quality attribute	Recreational and aesthetic benefits of residential properties close to wetlands[45] and lakes;[124] value of aquifer water storage for agricultural supply[125]

Continued.

Table 1. *Continued*

Technique	Approach	Limitations	Quality Checks/Indicators	Selected Examples
Replacement Cost	Estimates the value of a change in a nonmarket ecosystem service by calculating the cost of replacing the lost or reduced service with a manmade substitute or with restoration of the ecosystem	• Tends to overestimate • Few studies verify conditions necessary for validity • Can not estimate nonuse values	• Assessment of extent to which man-made replacement and lost ecosystems are substitutable and any significant differences in quantity and quality taken into account • Evidence that chosen replacement is least cost way of replacing—otherwise overestimate • Evidence that public are willing to pay for replacement costs (not necessarily a full-blown stated preference (SP))	Seed dispersal service of natural pollinators;[126] value of coastal protection and stabilization by mangroves;[127] yield increase attributable to aphid predation by natural enemies;[128] provision of clean drinking water by Catskill Watersheds[129]
Defensive Expenditure Method	This approach considers the costs and expenditures incurred in avoiding damages of reduced environmental functionality	• Issues relating to degree of substitutability • Typically lower bound estimate • Difficulty of disentangling value estimates when joint products provided • Can not estimate nonuse values	• Assessment of degree of substitutability, ideally goods will be prefect substitutes (or very high degree of substitutability) • Examination of perceived versus objective level of protection offered by substitute • Estimation of demand function	Coastal wetlands for hurricane protection;[130] storm protection services;[113,131] water purification benefits of forest watershed protection;[49] biogeochemical function of estuary[132]

Continued.

Table 1. *Continued*

Technique	Approach	Limitations	Quality Checks/Indicators	Selected Examples
Contingent Valuation Method (CVM)	An SP technique which elicits public preferences by *directly asking people how much they would be willing to pay (or accept) for a change in the quantity or quality* of a *given environmental good or service* in a hypothetical market[110]	• Time and cost in designing and implementing surveys • Loss of nontrivial information[38] • Noncompensatory decision strategies, e.g., warm-glow, rights-based • Problem of constructed, theoretically inconsistent preferences • Various sources of bias, e.g., hypothetical bias, strategic bias, insensitivity to scope, framing and elicitation effects	• Evidence of thorough and extensive pretesting of survey instrument: focus groups, cognitive interviews, and pilot testing • Inclusion of reminders of budget constraints and substitutes • Low rates of item nonresponse, protests and outliers (high rates may indicate weaknesses in scenario) • Model explanatory power, rejection of the null hypothesis that all coefficients on explanatory variables are equal to zero; and expected determinants of willingness to pay (WTP) are significant and correctly signed • Reasonable WTP estimates: WTP as proportion of income; consistency with other similar studies; and examination of confidence intervals • Assessment of tests incorporated for bias	Flood protection, recreation and habitat services of wetland;[133] habitat preservation for threatened species[134]

Continued.

Table 1. *Continued*

Technique	Approach	Limitations	Quality Checks/Indicators	Selected Examples
Choice Modeling	An SP method which elicits public preferences by asking respondents to *choose their preferred option from a series of alternatives*, each described in terms of its constituent attributes and levels	As for CVM, in addition: • greater cognitive burden may lead to random errors and difficulty in modeling responses. • Potential bias, e.g., inconsistency, learning and fatigue effects • Missing attributes • Technical complexities in design and data analysis	Similar to CVM, additionally: • All relevant attributes included, and levels are meaningful • For estimates to be welfare consistent, a baseline or opt-out option should be included unless in real-life a choice can not be avoided[135] • Attributes (and any interactions with socio-demographic variables) are significant and correctly signed • Where applicable the independence of irrelevant alternatives assumption should be met[92,136] • Confidence intervals for marginal WTP and overall welfare estimates • Assessment of tests incorporated for bias, e.g., inconsistency; survey satisficing; heuristics; dominant options and results assessed	Recreational, habitat and wildlife preservation functions of new wetland sites;[43] rainforest habitat preservation, and recreation;[137] preservation of wetland habitat and endangered species[138]

Continued.

Table 1. *Continued*

Technique	Approach	Limitations	Quality Checks/Indicators	Selected Examples
Deliberative Monetary Valuation	A hybrid of SP methods and discursive techniques of political science: small groups, selected to represent society, discuss and deliberate environmental issues in an open and fair environment	• Time and cost issues • May lack of representation due to small groups • Subject to group norms, e.g., dominant participants • May not result in monetary estimates • Possible bias due to recruitment process, e.g., self-selection bias	• Use of small groups • Selection process should ensure "society" is adequately represented • A fair and open structure/process for deliberation should be created (after Wilson and Howarth[82]): (i) each participant should be allowed to participate in discourse; (ii) each participant should be allowed to place issues on the agenda; (iii) each should be allowed to introduce his or her own assessment of an ecosystem good or service; (iv) each should be allowed to express their own attitudes, needs and preferences for an ecosystem good or service; and (v) no speaker would be hindered by external compulsion or pressure • Use of experienced moderator/ facilitator	River water quality improvements;[90] enhancements to floodplain habitat and wildlife;[85] improved river ecology, water quality and regulation;[88] wild good conservation;[87] and improvements to habitat, water quality and functions in Tillamook Bay estuary[139]

Sources: Refs. 11,91,106,114, and 122.

price. All attributes were found to be significant determinants of choice. Importantly, the distance attribute was negatively signed, indicating that utility declines as distance from the site increases—the so called "distance decay effect." This meant that assuming a constant unit value across populations for a specified change in ecosystem service provision would have led to biased estimates. Correspondingly, aggregate WTP was estimated using a spatially sensitive valuation function. This was operationalized using distance bands at 8-, 15-, 23-, and 32-mile intervals from the proposed site—WTP was calculated by multiplying the mean household WTP for each band by the total population within the band. By incorporating the "distance decay" effect, Luisetti *et al.*[43] were able to sensitize aggregate benefits to the socio-economic context. Even so, as the authors point out, additional variations in aggregate benefits would be expected if the proposed wetland creation occurred in an altogether different location—for example, where the adjacent population was larger, estimates would be higher; and in areas with a greater number of available substitutes, benefits would be expected to be lower. In both examples, a critical aspect of the valuation approach was the use of a geographical information system (GIS), which is emerging as a valuable tool in valuation. It is anticipated that the incorporation of spatial factors in ecosystem valuation is likely to become easier and more commonplace as access to GIS software and expertise increases.

Marginality

Eliciting marginal values can be technically demanding, and represents a challenge for nonmarket valuation in general, not just ecosystem valuation. To date, few studies have undertaken true marginal analysis of ecosystem transitions.[24,26,44] Mahan *et al.*,[45] for example, produce marginal value estimates of the value of wetland amenities to properties in Portland, Oregon. The results indicate that a property's value increases by $24.39 per *one-acre* increase in the size of the nearest wetland. Maler *et al.*[20] explicitly undertake marginal analysis in estimating the accounting price for the habitat service provided by a mangrove ecosystem to a shrimp population. Their model evaluates changes to fisherman well-being for a *10-hectare* change in the stock of a mangrove forest of *4000 hectares* in size, obtaining an accounting price of $200/hectare. In most cases, the ecosystem valuation literature has focused on valuing the stock—for example, Peters *et al.*[46] estimate the value of nontimber forest product (NTFP) services based on a stock inventory. Or, the actual service flow is valued—for example, Godoy *et al.*[47] value actual NTFP service flows from a Central American rainforest, and Croitoru[48] estimates annual flow of NTFP benefits for the Mediterranean region, while Adger *et al.*[49] estimate the total economic value of Mexican forest services. In some cases these analyses have been placed in a context of "change" by drawing comparisons with alternative land use options. For example, Peters *et al.*[46] and Bann[50] compare commercial timber extraction and NTFP harvesting for forests in Amazon and Cambodia, respectively. Yaron[51] examines the total economic value of three service flows, sustainable forest use, small-scale agriculture, and plantation agriculture—from forested lowland in the Mount Cameroon region—from the perspective of global, national, and local stakeholders. Kramer *et al.*[52] use the production function method and remote sensing to estimate the flood alleviation benefits to farmers from protecting upland forests in eastern Madagascar. The environmental change context was simulated by considering a policy proposal to establish a national park. First, deforestation rates were estimated using remote sensing and then projected into the future based on historical trend data. The land use changes were then used to predict effects on flooding. The value of the predicted reduction in flooding brought about by the establishment of the Mantadia National Park was estimated in terms of the value of reduced crop losses. Results indicated that the flood alleviation benefit of establishing the park is $127,700.

Double counting

In a cost-benefit analysis of a U.K. coastal managed realignment policy, Turner *et al.*[53] avoid the double-counting trap by treating the environmental benefits provided by the creation of intertidal habitats as a composite value. The authors used an estimate of £621/ha/yr based on a recent meta-analysis.[54] This value was assumed to incorporate the nutrient storage function (for nitrogen and particle-reactive phosphorus) on the basis that this provided an intermediate service to the final benefit of enhanced amenity and recreational quality. Farber *et al.*[55] similarly note the problem of including aesthetic services and nutrient regulation in a case study of Plum Island coastal ecosystem. In another example, Anderson *et al.*[56] explicitly address the double-counting trap by disregarding intermediate services in their benefits aggregation of the forest ecosystem service in the Brazilian Amazon. These examples are among a relatively small number in the valuation literature which directly seek to address the double-counting issue.

Nonlinearities

Barbier *et al.*[34] have stressed that for some ecosystems, for example, coastal mangroves, salt marshes, and other marine ecosystems, the services provided change in a nonlinear way as habitat variables, such as size of area, alter. They claim that recognizing such nonlinearities opens up the choice set available to policy makers. In the case of mangroves and the storm-buffering service they provide, it is argued that the nonlinear supply of the buffering service (i.e., reducing as successive landward zones of the mangrove forest are crossed) means that some mangrove conversion (e.g., to provide space for shrimp ponds) can be economically justified in cost-benefit terms. The authors note that an "up to 20%" conversion rule seems to be an emerging policy principle. But such generalizations are dangerous because ecosystem services must be assessed in a spatially explicit manner and with due regard for uncertainties surrounding possible threshold effects. In the mangrove example, the shrimp-pond locations and the current degradation status of the mangrove forest are crucially important. If the shrimp ponds are located on the seaward edge of the mangroves, they will be prone to storm damage and lost productivity. If the mangrove has already experienced significant degradation, it may be at or close to a threshold tipping point. Finally, mangroves (and other ecosystems) supply a range of interconnected services, the value of which needs to be included in any economic benefit and loss account.

Threshold effects

The challenge in incorporating threshold effects in ecosystem services valuation lies in our relatively limited knowledge of ecosystem complexity and interrelationships. Moreover, individual valuation studies frequently do not have the resources to undertake complex biophysical modeling. Consequently, the importance of threshold effects is often acknowledged in the valuation literature but rarely explicitly incorporated.[57] Soderqvist *et al.*[58] apply the travel cost method to value the benefit of a bigger fish catch to recreational fishers in the Stockholm Archipelago. The results indicate that doubling the average spring catch per hour of Perch from 0.8 to 1.6 kg amounts to a WTP of 56 SEK per angler. While on the surface this appears to be a small change, appropriate for marginal analysis, it is possible that the cumulative effect of doubling fish catch per hour could result in flipping the recreational fishery into an alternative state. Indeed, the authors note the need for further work in order to assess the potential effects (and costs) of measures that would improve fishing conditions in the archipelago. In a different example, Hein[59] explicitly incorporates threshold effects in modeling the optimum eutrophication control for a shallow lake ecosystem. Information on the supply of ecosystem services, the costs of eutrophication control measures, and the response of the lake to reduced nutrient loading (including the threshold effect) was combined in one ecological-economic model, to calculate the net benefit of eutrophication controls for the four biggest lakes in De Wieden wetland, Netherlands. Uncertainty regarding the point at which a switch to a clear water system occurs (the threshold) was incorporated via a sensitivity analysis. Threshold values were found to have a significant impact on the analysis.

Evaluating limitations and quality in ecosystem valuation

The standard assumption of mainstream economics—that people behave as rational self-interested individuals—has been undermined (since the 1970s) by scientific findings derived from the work of behavioral and experimental economists.[60] The current financial crisis has served to vividly highlight the shortcomings of the orthodox economic approach to public, including environmental, policy. Contrary to the neoclassical axioms, behavioral economics study findings suggest that individuals do not possess consistent preferences over all combinations of private and public goods and that these preferences are not reasonably stable across time and independent of the contexts in which, and the mechanisms through which, they are revealed. Rather, to a greater or lesser extent, preferences are endogenous—they change depending on personal experiences, social contacts and context, historical/cultural background, and the type of decision-making process.

Behavioral economics work has shown that people are prone to systematic biases when making economic decisions. They can opt for a course of action that is not apparently beneficial to themselves. They can value the same object more highly if they own it; they are overly influenced by the first option/suggestion in a list with which they are presented, and they can be conditioned by contextual details regardless of their relevance. When individuals try to decide the value of something, they tend toward the valuations that everyone else comes up with—a sort of "herd mentality."[61]

Intertwined with the technical debate over valuation theory and methods is a dispute over more ethical matters concerning the "proper" extent of markets ("commodification") and related behavior into environmental domains. In addition, where "isolated" individual preferences are the correct ethical basis for guiding social decision rules, rather than collectively influenced "citizen" preferences or more paternalistic collective preferences imposed by the "executive" to meet the "needs" of a civilized society.[8,62] The policy process can be steered by presenting people with all the options but with one given a more favorable treatment to "nudge" the decision into a predetermined "best social" outcome—"libertarian paternalism."[63] Some nudges work because they provide information about how citizens more generally behave in the same situation.

A sense of fairness and common purpose which prevails over selfishness is also a common finding in behavioral experiments. Trust seems to be a vital component of human interactions, including market transactions. Smarter cost-effective regulatory controls will be required to, among other things, restore levels of trust in transactions and environmental policy.

SP valuation methods, contingent valuation, and choice experiments have been at the center of the debate over the applicability of economic valuation. It seems incontrovertible that the behavioral economics findings have at the very least restricted the scope of the rational economic model. Some experimental economists have been testing the proposition that "anomalies" (in their terms) can be reduced by better survey/choice experiment designs, or via suitable arrangements, such as repeated markets and learning through experiences.[64–66] Other analysts view the behavioral evidence as support for alternatively held preference theories. Thus, the existence of "protest bids" in survey when respondents refuse to make tradeoffs between an environmental good/service and money may indicate the presence of lexicographic preferences often linked to rights-based beliefs.[67–70] Tversky and Kahneman[71] have provided another angle called the "reference-dependent" preference or "endowment" effect. This does seem to provide a plausible explanation for another "anomaly"—the willingness to accept (WTA) versus WTP disparity.[72] Using this approach in cost-benefit analysis, Knetsch[73] argues that WTA loss is two to four times more than otherwise commensurate gain WTP and that WTA is the appropriate reference state for environmental losses.

A wider question follows from the more specific debate over valuation methods: What is the appropriate "scope" of cost-benefit analysis? Issues related to incommensurability and incomparability now take center stage. The process of reducing environmental systems to a single or aggregated monetary value has been criticized due to the loss of "nontrivial" information on other important dimensions of value.[38,74] It has also been argued that money is not a neutral scale of valuation with no value *per se*, nor is it universally substitutable.[75] So, faced with a "large" environmental loss relative to the status quo, and assuming lexicographic preferences or a reference-dependent effect, an individual may not be compensated by a sum of money. However, compensation may be acceptable on a "like for like" or "in-kind" or social capital substitute basis.[75–77]

Endogenous preference theories (i.e., involving dependence on an individual's personal history, collective interaction with other citizens and social context) together with endowment effects and "other regarding" preferences seem to fit more comfortably into the prevailing and emerging environmental policy agenda. Individuals may act to affect the welfare of others—they can make different decisions as citizens rather than consumers, in isolation or in a collective social context, and the process by which decisions are made (e.g., is it fair?) may be influential.[60,78,79] Preferences can be socially conditioned or conditioned by cultural transmission in line with prevailing norms and patterns.[80,81] So, survey methods that use citizen frames to elicit values or more deliberative forms of evaluation do offer the promise of new policy-relevant information.[82]

In response to some of the criticisms associated with standard economic valuation approaches, a new set of deliberative approaches has emerged which combine SP methods with the discourse-based techniques of political science.[83] These methods involve small groups usually of between 5 and 20 individuals, who are selected to represent "society" and are provided with information about a particular environmental policy question, which they are asked to discuss and formally deliberate in an open and fair environment. By engaging in group discussion, individuals are believed to be exposed to a much richer set of information, attitudes, and experiences, enabling a much better understanding of the issue at hand. In comparison with standard SP studies, the method allows more time for processing and assimilating information. Spash[84] describes the process as being a *transformative* experience; as such, it moves away from the standard economic assumption of preformed and stable preferences, which have been questioned by recent empirical evidence, to embrace preference construction as part of the process, and which may lead to better-informed and more stable values and judgments.[83,85] The deliberative component is also thought to encourage individuals to extend beyond their own personal welfare so that the resulting values, judgments, and outcomes will reflect a more complete and socially equitable assessment of the issue at hand.[82,86] This so-called *moralizing process*[84] may enable better representation of invisible groups, such as future generations or nonhumans, which can be facilitated through the use of witness evidence.[83] In practice, the extent to which social values are represented depends on the specific approach and institutional context—some studies simply elicit individual WTP which has been informed by group debate,[87] while others ask individuals or groups to respond specifically in terms of what is best for society.[88] Studies have resulted in a variety of distinct values. Spash[84] provides a useful categorization differentiating between values associated with individuals, which may be exchange values, charitable contributions or fair prices, and social values, which may be speculative, expressive, or arbitrated (p. 696). In terms of outputs, the open and discursory nature of the technique generates a much deeper understanding of stakeholder attitudes toward policy options, which offers an avenue for expressing noneconomic measures of value[83] as well as monetary measures. Where monetary values are elicited the latter are thought to offer advantages over standard SP methods by extending beyond personal welfare, to better reflect the goals of social equity and political legitimization. However, as Spash[84] points out, studies which seek to constrain deliberative monetary valuation (DMV) to be as close as possible to conventional SP—for example, by excluding responses motivated by "fairness" due to possible strategic behavior,[87] may lose some of the key benefits intended by the approach (i.e., the inclusion of social values, rights, and equity). Finally, the method also has the added value of incorporating stakeholder participation in the policymaking process, which in itself may be regarded as welfare enhancing.[74]

While deliberative valuation methods hold much promise, many of the hypotheses surrounding their benefits have yet to be fully tested.[82] A number of limitations can be envisaged. First, (as with SP) the method may be costly to implement and can take a considerable amount of time—for example, individual citizen juries may take several days to deliberate. The use of small groups may make it difficult to claim representation[84] (where this is part of the objective), and to apply the resulting values to the wider population.[89] There may also be additional bias due to the recruitment process and possible self-selection.[90] The deliberative process may be subject to group norms—for example, dominant participants may unduly influence proceedings/other individuals, others may express polarized views that do not truly reflect their beliefs, and shy individuals or minority view holders may fail to fully participate.[83,89] There is an implicit assumption that all individuals will be able to effectively communicate their opinions and beliefs, which may result in exclusion on the basis of poor education or poor verbal communication skills.[84] Group discussion may not be the appropriate medium for eliciting attitudes and preferences for sensitive issues despite attempts to create a free and open environment. It has been found that groups do not always pool their unshared information.[82]

From a valuation perspective, there is still the opportunity for strategic behavior, and it is not necessarily the case that DMV will result in a monetary value, either by design[85] or because individuals may still fail to tradeoff. Finally, results may be unintentionally influenced by the moderator/facilitator.

DMV is an emerging tool in ecosystem valuation, but as of yet there is no clear guidance on how to assess quality; however, lessons may be learned from the literature on the use of focus groups in valuation and on political discussion techniques. Some necessary conditions can be envisaged. Small groups should be used, comprising at least two individuals and less than 20. The selection process should ensure that "society" is adequately represented. A fair and openly structured procedure for deliberation should be implemented. Wilson and Howarth[82] recommend: (i) each participant is allowed to participate in discourse; (ii) each participant is allowed to place issues on the agenda; (iii) each is to introduce his or her own assessment of an ecosystem good or service; (iv) each is allowed to express their own attitudes, needs, and preferences for an ecosystem good or service; (v) no speaker is hindered by external compulsion or pressure; and (vi) the goal of discourse would be to reach a consensus value among the participants. Finally, it is essential that an experienced moderator/facilitator is used. For further guidance on DMV, see Spash.[83]

Given the limitations we have just reviewed and the on-going nature of elements of the debate over valuation (and cost-benefit analysis), it is still the case that an array of economic valuation techniques and results exist which are or may be used to value ecosystem services. While these have been extensively applied to the valuation of ecosystem service flows, they sometimes lack sufficient rigour.[15] Correspondingly, prior to using the results of a valuation study, it is important to evaluate its internal quality. In the appendix we describe some generic quality checks that may be applied and provide more tailored guidance on how to assess validity and reliability with respect to specific valuation techniques.[91]

Benefits transfer is an approach to valuation which uses information about benefits[c] captured at one place and time (the "study" site) to estimate the economic value of environmental goods and services at another place and time (the "policy" site).[92,93] This is achieved either by transferring an adjusted or unadjusted unit WTP value—for example, mean or median WTP—or by transferring an entire valuation function that describes the relationship between WTP and explanatory variables. The key attraction of the technique is that it reduces the need for original valuation studies, thereby offering significant cost and time savings for any study seeking to estimate the value of a nonmarket environmental good. However, the validity of the technique itself remains open to question, and in many cases the results have been far from satisfactory. Given the spatially explicit characteristics of ecosystem services the benefits transfer tactic requires heavy scrutiny.

The main limitations of the technique concern the various errors which arise in its application, namely: (i) measurement error; (ii) transfer error; and (iii) publication error.[94] Measurement error arises due to errors in the primary studies typically due to problems with survey design, unrepresentative sampling, poor application of econometric methods, low explanatory power of model, and so on. The quality of the primary study inevitably affects the accuracy of any subsequent value transfer.[93–96] The second source of error occurs due to the process of generalization and is a reflection of the level of dissimilarity between the study and policy sites in terms of the population, physical site characteristics, environmental resource, and welfare measures. It is of particular concern as sizeable transfer errors have been reported in the literature. For example, Bateman *et al.*[92] observe that errors are between 1–75% if outliers are excluded but are up to 450% if included.[d] The acceptability of these error margins depends very much on the purpose of the benefits transfer study. A lack of policy site data on nondemographic determinants of WTP, such as attitudes, may compound the problem.[97] The final source of error relates to an apparent predilection for original valuation studies to focus on sites that are *a priori* considered to be valuable. If this is the case then

[c]Or costs, in which case the term "value" transfer is more appropriate.

[d]Transfer errors may be calculated from the following formula[106]: Transfer error = [(transferred estimate − original study estimate)/(original study estimate)]*100.

the resulting benefits transfer can be expected to be overestimates. In particular, Hoehn[98] finds that, in the context of ecosystem services, original valuation studies are dominated by those focusing on ecosystems with higher profiles or more social benefits. Additionally, the technique is limited by more inherent problems, such as its lack of capability to deal with unique resources and issues relating to temporal stability.

Although set procedures for benefits transfer do not yet exist, some general guidance is available on the basic conditions necessary for valid transfer. Error seems to be minimized the more similar the sites are with respect to physical and market characteristics.[94] Although Ready *et al.*[99] found no significant difference in transfer error between the transfer of unadjusted, adjusted, and the benefits function, in general, accuracy seems to be higher where benefits functions are used rather than transferring unit values.[92] Accuracy is also found to be higher where transfer is based on a large sample of original studies.[92] Indeed, Johnston *et al.*[100] find that studies based on one or two original studies are found to introduce substantial uncertainty and broader error bounds. Spash and Vatn[97] recommend greater account is taken of a wider range of explanatory variables—for example, by placing more attention on adjusting for social and attitudinal variables for market consistency. Although incorporating data on attitudes may necessitate the collection of primary data at the policy site, Brouwer[101] argues that the additional cost may be justified by the increased validity of the results.

Conclusions

In this chapter we have discussed the necessary conditions for taking ecosystem valuation forward so that it is commensurate with scientific information and the emerging findings from behavioral and experimental economics, as well as being meaningful from a policy perspective. To achieve this we have argued that it is useful to consider a sequential analytical process which encompasses: (i) the spatial context of ecosystem service provision and beneficiaries; (ii) appropriate application of the concept of marginal analysis; (iii) avoidance of the double-counting trap; (iv) as far as is feasible a comprehensive understanding of the underlying biophysical relationships so that nonlinearities may be identified; and (v) full consideration of possible threshold effects. To quality assure the economic analysis within this framework, the valuation techniques themselves need to be appropriately chosen, configured, and applied; this requires consideration of generic quality issues, such as the overall study design, data quality and robustness of statistical analysis, and the validity and reliability of results and aggregation procedures, as well as specific quality indicators related to the individual valuation technique. Finally, benefits transfer procedures should be treated with great caution despite their superficial cost-effectiveness advantages from the policymakers" perspective. Advances in GIS and related techniques may offer some prospects for improved benefits transfer in a limited number of instances.

Overall, we conclude that there is a legitimate and meaningful role for regulated market transactions and related human behavior in the environmental domain and that therefore cost-benefit analysis is not ruled absolutely out of scope. For a critique of this position see, among others, O"Neil,[102] Burgess *et al.*,[103] and Aldred.[75] We also believe that a typology of environmental values based on "use" and "nonuse/existence" value categories, more or less captures human related instrumental and intrinsic values of nature.[104] Comparability and monetary incommensurability problems are significant but not totally intractable, at least as far as "use" and some but not all "nonuse" values are concerned. Other economic revealed preference valuation techniques can provide "pricing" value information on a number of environmental goods and services. What the behavioral economics (and related disciplines) findings do require is a reappraisal of the proposition that, since environmental public goods often do not have market prices and society requires that their provision should be decided by collective decision, it is legitimate to take into account individual preferences, values, or attitudes and guard against the dominance of special interest groups and/or special pleading.[105] While it is the case that the aggregation of individual preferences represents the collective choice outcome, it is far less clear that the individual preferences themselves are given and context independent. The opposite is often the case in the environmental policy context and preference formation and elicitation processes are key. Collective rather than individualistic social arrangements for valuation seem to be the direction in which the behavioral economics findings are pointing.

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Conflicts of interest

The authors declare no conflicts of interest.

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Appendices

Box 1

Generic Quality Checks for Ecosystem Valuation Studies

Overall Study Design[91]

First, when evaluating the quality of a valuation study, consideration should be given to the adequacy of the overall study design—that is, whether the study aims are reasonable and whether it measures what it intends to measure. For example:

- Has an appropriate valuation been used (e.g., only SP methods can capture nonuse)?
- Has the correct welfare measure been used taking into account property rights and the nature of environmental change[107]?
- Is the valued change realistic and relevant (e.g., has a clear causal relationship been established between the change in service and well-being)?
- Are the study assumptions reasonable?
- Has the study been subject to external review?

Data and Statistical Analysis

An assessment should be made of the quality and appropriateness of the data used—this is often a key constraint in ecosystem services valuation[15]—and, of the analytical methods adopted. For example:

- In general primary data is preferred since it has been generated for the specific study purpose, in which case data should be collected in a reliable and appropriate way.
- Has the appropriate biophysical data been collected for measuring the ecosystem service?
- Are samples based on appropriately defined target populations, sampling frame, and sampling methods? Are they sufficiently large?
- Is best practice followed in data collection (e.g., Dillman's Total Design Method for mail surveys[108])?
- Is the statistical analysis appropriate (e.g., correct assumptions for functional form)?

Validity and Reliability of Estimates and Aggregation Issues

A decisive element in the evaluation of any valuation study is the extent to which the resulting welfare estimates are considered *valid* and *reliable*. Validity refers to the degree to which a study measures the intended quantity and reliability is the degree to which the estimates are stable and reproducible.[92,109] For example:

- A simple "sanity check": Do the results seem reasonable[15]?
- Expectations-based validity: Do the results conform with theoretical expectations and other empirical results[110]? Does the econometric function indicate expected relationships?
- Convergent validity: How do estimates compare with those derived from other techniques?
- Reliability: Does the estimated econometric model have satisfactory explanatory power (e.g., for CVM studies, does the model have an r^2 greater than 0.15[110])? Do test-re-test procedures indicate stability?
- Has aggregation been appropriately conducted (e.g., adjustments for unrepresentative samples) and is double-counting absent?

Adapted from: Soderqvist and Soutukorva.[91]