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Beyond monetary measurement: How to evaluate projects and policies using the ecosystem services framework

Frans J. Sijtsma [a,*], C. Martijn van der Heide [b,c,1], Arjen van Hinsberg [d,2]

[a] Faculty of Spatial Sciences, University of Groningen, P.O. Box 800, NL-9700 AV Groningen, The Netherlands
[b] Natural Resources Division, LEI Wageningen UR, P.O. Box 29703, NL-2502 LS The Hague, The Netherlands
[c] Van Hall Larenstein, University of Applied Sciences, P.O. Box 9001, NL-6880 GB Velp, The Netherlands
[d] Netherlands Environmental Assessment Agency (PBL), P.O. Box 303, NL-3720 AH Bilthoven, The Netherlands

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ABSTRACT

In this paper we focus on how to achieve better decision support when decision-makers use the ecosystem services (ESS) framework to broaden their evaluations. We contribute to the debate on valuation of ecosystem services by inquiring into how the ESS framework relates to the judgement and measurement provided by Cost-Benefit Analysis (CBA) and Multi-Criteria Analysis (MCA) evaluation techniques. We argue that Multi-Criteria Cost-Benefit Analysis (MCCBA), which is a carefully designed combination of CBA and MCA, provides a good starting point for the evaluation of projects or policies involving changes in agricultural and natural ecosystem services.

The main characteristic of this MCCBA approach linked to ESS framework is its threefold evaluative endpoint structure to account for (i) basic health, (ii) economic welfare, and (iii) higher well-being. The third endpoint includes concerns about the well-being of nature. The MCCBA approach utilises highly standardised cardinal or ratio scale measurements, in particular we use two existing measurements, known as Disability Adjusted Life Years for basic health, and monetary Net Present Values for economic welfare. We also introduce one new measurement: Threat weighted Ecological Quality Area to account for nature's well-being. We argue that evaluation of projects or policies involving many different ecosystem services should use these three endpoint measurements.

1. Introduction

Increased agricultural productivity has over time facilitated economic development in which larger and larger urban concentrations play a pivotal role (McCann and Acs, 2011; Strijker, 2005). One could even say that increased agricultural productivity has facilitated the development of a socio-economic system 'away from nature' (Buijs et al., 2010). And although high productivity increases in agriculture, as in forestry and fisheries, build on natural processes and conditions, they too seem to shift agriculture 'away from nature', since agriculture faces an increasingly tense relationship with biodiversity and ecology (Björklund et al., 1999; Stoate et al., 2009). The ecosystem services (ESS) framework, as highlighted by other contributions to this special issue,

* *Corresponding author*. Tel.: +31 50 363 37 54.
E-mail addresses: f.j.sijtsma@rug.nl (F.J. Sijtsma), martijn.vanderheide@wur.nl (C.M. van der Heide), Arjen.vanhinsberg@pbl.nl (A. van Hinsberg).
[1] Tel.: +31 70 335 82 18.
[2] Tel.: +31 302743062.
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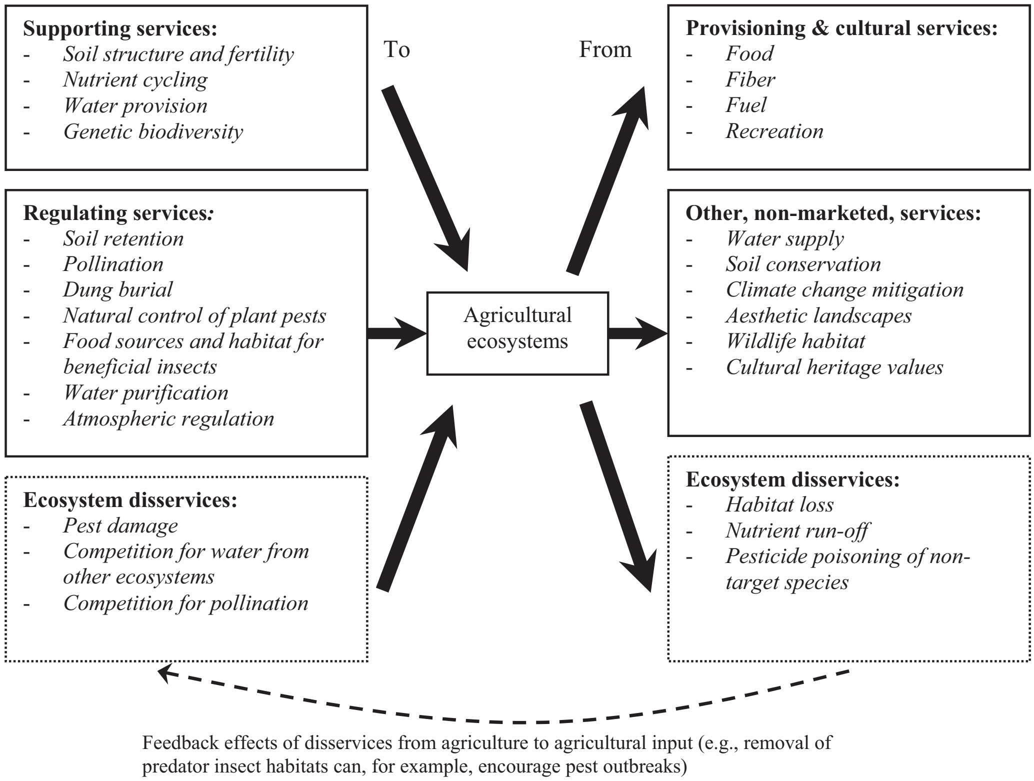

Fig. 1 – Ecosystem services and disservices to and from agriculture.
***Source:* Zhang et al. (2007).**

denotes the benefits that people derive, directly and indirectly, from nature (Turner et al., 2010). In a way, the ESS framework can be seen as a means of reconnecting urban and agricultural systems to nature, by informing decision-makers of the many and complex interrelations between these systems and nature.

The authoritative *Millennium Ecosystem Assessment* (MEA, 2005) distinguishes 30 ecosystem services[3] which specify these links between nature and human well-being and assigns them to four distinct categories: (i) provisioning services, such as the production of food, timber, fibre, and water; (ii) regulating services, such as the regulation of climate, floods, and disease; (iii) cultural services, such as knowledge, spiritual and recreational benefits; and (iv) supporting services, such as nutrient cycles, soil formation and crop pollination. Zhang et al. (2007) depict a more detailed picture of 27 services related to agriculture that also includes six disservices (Fig. 1).[4] If we consider farm level management options (Ribaudo, 2008), this picture becomes even further elaborated.

Significantly in support of our aim is that the ecosystem service framework is designed to assist decision-making (Fisher et al., 2009; MEA, 2005). Decision-making typically involves a choice between alternative project variants or policy options, say, A, B, and C to X in Table 1 (Belton and Stewart, 2002). Deciding which option is best requires an evaluation of the different impacts of the policy options. Basically, the ESS framework broadens the scope of evaluations by encouraging decision-makers to consider a wider range of impacts and thus a larger number of impacts. If a decision-maker who would normally consider a certain set of policy options (Table 1: A, B, and C, to X) and a certain set of impacts (1, 2, and 3 to Y), were to also use the ESS framework, this implies that the set of Y impacts under scrutiny in the decision process is enlarged to Y *plus* the amount of ESS considered. For example, a farmer who needs to decide on a new crop might normally consider impacts on, say, his income, future market possibilities and daily workflow; however, using the ESS framework would also alert him (see Zhang et al. (2007) to impacts on pollination, natural control of plant pests, water purification, etc. Likewise, a regional agricultural policy maker deciding on a new subsidy scheme for small farmers might normally consider, say, number of farmers affected, impact on their living standard, erosion impacts, and changes in land ownership; however, using the ESS framework would stimulate him to consider, with MEA, the impacts of the new scheme on a broader range of regulating services (i.e., climate regulation, waste treatment, disease regulation, etc.) as well as cultural services (impacts on cultural diversity, spiritual and religious values, aesthetic values, social relations, cultural heritage values, and recreation). If the decision-maker follows Zhang et al. (2007), there may be 27 ESS; if the MEA is followed there may at least

[3] Without claiming to be complete. We therefore sometimes speak of '30+' ecosystem services.

[4] Zhang et al. limit cultural services to recreation and rank the others under the heading of *other, non-marketed services*. Like MEA, their list is more illustrative than complete.

Table 1 – A basic decision-making scheme and the consequences of the ESS framework.

		Policy or project options considered by the decision-maker:					
		A	**B**	**C**	**…**	**…**	**X**
Impacts evaluated by the decision-maker:	**1**	a1	b1	c1	…	…	x1
	2	a2	b2	c2			x2
	3	a3	b3	c3			x3
	…						
	Y	ay	by	Cy			Xy
	ESS 1	aess1	bess1	cess1			xess1
	ESS 2	aess2	bess2	cess2			xess2
	…						
	ESS 27	aess27	bess27	cess27			xess27
	…						
	ESS 30+	aess30+	bess30+	cess30+			xess30+

Evaluation methods

Commonly either Cost-Benefit Analysis or Multi Criteria Analysis

	A	B	C	…	…	X
Single score	az	bz	cz			xz

be 30 ESS (see Table 1). The decision-maker has to take into account the ESSs that are relevant and new. Obviously not *all* ecosystem services must be new to the decision-maker; there may be some overlap between the 1 to Y impacts and the ESS 1 to 30+ impacts. Nevertheless, due to the stance of the ESS framework of reconnecting to nature, the aim is to give greater attention to commonly under-represented or disregarded links between nature and human well-being; the ESS framework will generally imply a broader range of impacts to be considered.

Evaluation first involves calculating scores to fill the cells (a1 to xess30+ in Table 1) measured in their natural units (tons, Euros, meters, etc.). Evaluation methods then support decision-making by somehow adding up scores of the considered impacts into a more compact score, commonly often a single score for each policy or project option (az to xz, in Table 1; see below for how CBA and MCA do this). In the process of reaching a more compact score, the natural unit measurements have to be brought to a common measurement scale (Sijtsma, 2006).

Although the ESS framework has great potential for improved decision-making, in our view this potential can only be realised if the evaluative structure is analytically sound, and the accompanying empirics of decision support are standardised (Kontogianni et al., 2010) and easy to use and understand (Cowling et al., 2008). In their absence, however, we think that the ESS framework could produce confusion due to the interrelatedness of many impacts, and may generate an unwieldy multi-disciplinary research agenda due to many new (but not yet) fully documented impacts, and therefore not improve on decision support (Wallace, 2008).

In this paper we focus on how to achieve better decision support when decision-makers use the ESS framework to broaden their evaluations. This topic has been under considerable debate for some time (Carpenter et al., 2009), a major issue being the amount of monetisation of ESS that is achievable when we consider all ecosystem services (Braat and ten Brink, 2008; Clark et al., 2000; Costanza et al., 1997; Sukhdev, 2010), or mainly agricultural ones (Dale and Polasky, 2007; Porter et al., 2009). This monetisation debate is strongly related to the use of Cost-Benefit Analysis, but decision support using ESS is not limited to CBA. Multi-Criteria Analysis (MCA) is also a very popular evaluation tool for ESS related decision support (MEA, 2005; Slootweg and Van Beukering, 2008). This paper in Section 2 considers the merits of both CBA and MCA in handling the added complexity due to the shaded area of Table 1, and argues for the use of a mixed approach (MCCBA) to provide solid decision support in this setting. This MCCBA approach implies working with a threefold division in well-being domains, and three essential standardised measurements. Two of these measurements already exist, while the third concerns a new measurement of nature well-being:

Threat Weighted Ecological Quality Area (T-EQA), discussed in Section 3. Finally, Section 4 discusses key aspects related to the future research agenda.

2. A sound evaluative basis

2.1. Introduction

To facilitate the trade-off among (competing) goals and to evaluate the wide ranging impacts project might have on ecosystem services, such as a change in cultural heritage values or changes in pollination, a variety of evaluation tools can be used. Cost-Benefit Analysis (CBA) and forms of Multi-Criteria Analysis (MCA) are the two most commonly employed tools capable of responding to this concern.

2.2. CBA and ESS

Cost-Benefit Analysis takes as its starting point the preferences of individuals with regard to proposed changes (Boardman et al., 2011; Hanley and Barbier, 2009; Mishan and Quah, 2007; Pearce et al., 2006). Utilitarianism forms the foundation for CBA, which apparently does not conflict with the ESS approach. As van Kooten (2000, p. 147) notes, ''[u]tilitarians consider only the instrumental values of biodiversity ...'' and the ESS approach does just that.

In its design, CBA is closely connected to what is known as an 'endpoint approach' in the ESS debate (Chapman, 2008; Kontogianni et al., 2010). In CBA all impacts are assessed with the welfare of the affected individuals as its aim. It is geared towards marginal changes in welfare from the given situation (Cowling et al., 2008). In its function, CBA employs monetary Net Present Value (NPV) calculations to determine whether a certain decision or action will result in a net benefit or a net loss to society. Although focussed on individual preferences, the flexibility of CBA and the utilitarian ethics underlying it permit the suppression of individual preferences for the good of the 'collective', that is, wider society, as long as the gain exceeds the loss (van Kooten, 2000).

In its measurement, CBA relies largely on the function of markets; this is where its monetary stance comes from. Using market outcomes facilitates the work of adding welfare changes because – provided that the market functions efficiently – the market carries out much of the adding up for the evaluator. However, of the 30 ESS that we count in the MEA, only a mere 25% of them are measured readily in market-related monetary terms. If the market cannot handle the valuation, the CBA analyst needs to calculate comparable values. Therefore, since lists of ecosystem services are always lengthy, and only about one-quarter of them are market-related, the combined use of CBA and the ESS framework thus calls for a Herculean monetisation effort, as can be observed in major ESS studies (Björklund et al., 1999; Braat and ten Brink, 2008; Costanza et al., 1997; Porter et al., 2009). In fact, economic valuation has become the battleground for oppositional notions of how economics should be practiced (Clark et al., 2000; Sugden, 2005; Sijtsma, 2006).

The following observations about CBA and ESS can now be set out:

- Utilitarian CBA fits well with the ESS framework
- To work efficiently, CBA requires monetary measurement of all impacts
- Combined use of CBA and ESS framework provokes a very substantial monetisation research agenda, which is presently under fierce methodological debate.

2.3. MCA and ESS

Multi-Criteria Analysis (MCA) is also used to uncover the trade-offs between agro-ecosystem services that people are willing to make (Carpenter et al., 2009; MEA, 2005; Slootweg and Van Beukering, 2008). MCA takes as its starting point the preferences of a decision maker or group of decision-makers or sometimes a broader group of stakeholders relevant to a project (Belton and Stewart, 2002; Gamper and Turcanu, 2007; Pomerol and Barba-Romero, 2000). As a project or policy decision will have various different impacts, MCA measures these impacts as separate criteria. The criteria structure for an MCA assessment has to meet a number of conflicting demands (Keeney and Raiffa, 1976). On the basis of the scores on the different criteria, and the relative weights given to the criteria, the best choice can thereby be determined.

At the measurement stage an important intermediate step in any MCA is to make a so-called impact matrix, which shows the scores on all the impacts for all the considered alternatives. But for many decision-making processes the impact matrix is far too cumbersome. For instance, in the MEA study an unwieldy table of four and a half pages is presented for changes in all ESS (MEA, 2005, pp. 41–45). An overly large impact matrix provides too little judgment; therefore as is common in MCA, it is made more compact (Belton and Stewart, 2002).

An elaborate criteria table will normally have a tree-like hierarchical structure, with the massing of all criteria into groups (branches) and sub-groups (twigs). Adding the scores of different criteria requires scaling criteria and assigning weights (Hermann et al., 2007). For instance, in The Millennium Ecosystem Assessment the impact scores were scaled to +, 0 or −; whereas for the weights between criteria, a unitary weight – giving all criteria an equal weight – was chosen. The giving of weights is very subtle yet essential to MCA. However, for what follows it is crucial to distinguish between higher and lower order weights. Within a hierarchically ordered tree of criteria, weights for sub-criteria are known as *lower order weights*, an example of which may be the relative weight of say methane and nitrous oxide, which are added up in a so-called CO_2-equivalents, a measurement commonly used to assess the ecosystem service of climate regulation (Hertwich et al., 1997). Weights for the near final-criteria, which are high in the hierarchical order of criteria, are known as *higher order weights;* for example, the relative weight of *employment* versus *biodiversity*. One important lesson learned in MCA practice is that the higher order weights are often easily contestable in public debate, while many lower order weights, which are frequently provided by experts, may remain uncontested (Sijtsma, 2006).

The following observations about MCA and ESS can now be stated:

- Using the MCA methodology for ESS assessment will easily lead to an elaborate criteria structure of (over) 30 criteria

Table 2 – The general valuation approach of CBA, MCA and MCCBA.

	CBA	MCA	MCCBA
Judgement criterion	Welfare changes of the population experiencing effects: costs and benefits to whomever they accrue.	Preferences of decision makers or stakeholders concerning the changes on different criteria.	Well-being changes of the impact population(s) (as with CBA) including non-monetary measurement of health and biodiversity concerns of the population (building on MCA).
Measurement stance	Preferably monetary measurement. Costs and benefits over time aggregated using social discount rate.	Criteria measured in their own dimensions; aggregation requires scaling of criteria. Relative weighing of criteria by decision-makers or stakeholders.	Consensus-based measurement (monetary and non-monetary), thus avoiding easily contested higher order relative weighing.

- Adding these criteria requires scaling and weighting of the criteria. Scaling usually leads to measurement that follows the weakest form of quantification: most often +, 0 or − scores.
- Giving weights for higher order criteria is a complicated and thorny problem.

2.4. *MCCBA and ESS*

A mix of CBA and MCA may also be used. Some mixes loosely combine CBA and MCA, others are more developed and standardised. Cost-Utility Analysis (CUA), which is widely used in health care evaluation, is an example of a developed and standardised combination (Drummond et al., 2005; McPake et al., 2002, pp. 93–94).[5] CUA combines the standardised measurement of health impacts using Disability Adjusted Life Years (DALYs)[6] alongside monetised costs. However, Cost-Utility Analysis has been designed in particular for health evaluations, not for the ecosystem service framework. As Slootweg and Van Beukering (2008) show in their interesting overview of ESS evaluation cases, thus far only loose combinations of CBA and MCA are used in this field (in conjunction with the separate use of each technique).

For a broad-based technique in which CBA and MCA are combined in a standard and theoretically grounded way, we will turn to the MCCBA approach developed in Sijtsma (2006). Like Cost-Utility Analysis, the MCCBA approach uses standard measurements of health impacts alongside monetised costs and benefits, but unlike Cost-Utility Analysis, it is not limited only to health assessments: it can be applied to a wide range of assessment problems. As we will show below, the use of MCCBA for ESS assessments involves at the least adding non-monetary measurement of biodiversity impacts along with health impacts and monetised costs and benefits.

Table 2 outlines the general evaluation approach of CBA, MCA and MCCBA concerning the two essential dimensions: their judgement criterion and their measurement stance.

[5] Cost-Utility Analysis can be seen as a specialised and standardised form of cost-effectiveness analysis, and thus comes from the cost-benefit tradition (Boardman et al., 2011). The utility part of it is, however, closely connected to Multi-Attribute Utility Theory, which comes from, and may even be central to, the MCA tradition (Edwards and Newman, 1982; Jiménez et al., 2003).

[6] Or Quality Adjusted Life Years (QALYs).

2.5. *A deeper understanding of the evaluation problems*

The MCCBA approach (Sijtsma, 2006; Sijtsma et al., 2011) as a mixed method approach has as its point of departure the observation that CBA and MCA encounter continuous valuation and communication problems in providing formal decision support for projects or policies involving ESS. Straightforward CBA and MCA assessment results are therefore never beyond dispute. We see at least two fundamental and shared reasons for the problems which we sketched above when discussing CBA and MCA separately.

The first fundamental reason for the ongoing problems is indicated by the unclear mix of different spatial scales. As the MEA 30 ecosystem services form shows, environmental functions may operate at local, regional, national or international scales, and the provisioning service may function likewise, but from an evaluation-oriented measurement perspective the actually relevant spatial scale when filling the impact scores of Table 1 needs to be clear.

The second fundamental reason relates to different concepts of well-being (Gasper, 2010). In our view the distinction of cultural services within the ESS framework makes this issue most pressing and we will therefore discuss cultural ecosystem services and well-being in some depth.

Apart from the well-known recreation and leisure activities, the cultural ecosystem services include spiritual, philosophical, religious contentment, aesthetics, knowledge, and education (Wallace, 2007). The cultural services distinction within the ecosystem services is strongly grounded on Maslow's *hierarchy of human needs* (Maslow, 1948; Rowan, 1998; Wallace, 2007) in which cultural services are situated in the upper part of the needs hierarchy. The lower needs are, for instance, requirements for food and safety. Lower needs are known as *deficiency-dominated needs* in which individuals are motivated to overcome the discrepancy between their actual state and some optimal adequate state. For the higher needs, known as *growth needs*, context and circumstances differ markedly, because at this stage individuals lack final targets or optimal states (Heylighen, 1992).

Heylighen argues convincingly that no Maslow-inspired hierarchy of needs can ever be absolute; any class may only be a rough approximation. Furthermore, Heylighen's systems approach is extremely useful since it links the personal/individual needs level to the regions and countries level. Finally, and significantly to understanding CBA, MCA and

MCCBA's possibilities, Heylighen redefines Maslow's hierarchy of needs in terms of the *urgency* of (potential) perturbations experienced by the system. Urgent perturbations have a high probability of destruction and a short time horizon. Non-urgent perturbations have a weak probability of destruction, a long time horizon and high potential for 'growth' (Heylighen, 1992). Using Heylighen's interpretation, we may define a tripartite division of well-being, which is defined by two extremes: with basic well-being as completely urgent and higher well-being as completely non-urgent (see van Kooten's *high level ethical norms*, 2000). For lack of a better word, we will label the intermediate category between the two extremes as 'everyday' well-being. From an evaluation and measurement perspective, it is then fruitful to identify three well-being domains: *basic*, *everyday* and *higher well-being*.[7] This tripartite distinction is designated solely for the purpose of evaluation, i.e., in helping decision-makers cope with Table 1, since the specification of these domains highlights crucial characteristics of preferences, especially on the issue of possibilities for measurement and specifying judgement on trade-off relations (compare Table 2).

Hunger, thirst and physical safety correspond to an individual's basic health, and if they are deficient they will have the highest urgency. In this context there are no trade-offs, just dominance. Thus those impacts in Table 1 that directly affect basic needs will simply have a different priority than all others: a farmer struggling for survival will not care too much for the educational services his situation may provide. In contrast, the higher well-being domain, that of growth needs, is characterised by openness and learning. For higher order needs, there is less urgency to make clear distinctions, and what's more, it is more difficult to do so. ''The least 'urgent needs' correspond to completely ill-defined problems: if your goal is learning or exploration, then there is no criterion which tells you when you have achieved your goal.''[8]

For an ESS evaluation and MCCBA, opaqueness in the definition of aims that characterise a higher well-being domain does not mean the absence of measurement options. Quite the contrary. The implication here is that it is harder to find fixed and clear-cut higher order weights, that is, trade-off relations between the 1, 2, and 3 to Y to MEA 30+ impacts of Table 1 that relate to the higher well-being domain. For example, it will be hard to detect clear-cut trade-off relations stating how much money people are willing to offer to relieve world poverty or halt the loss of biodiversity (Turner and Fisher, 2008). Such are tough questions indeed. Nevertheless, poverty and the protection of biodiversity are clear concerns, and to specify, that is, to *measure*, exactly how much world poverty or how much biodiversity is at stake when choosing between policy options is important information. The major MCA thinker Ralph Keeney called it the most common and at the same time most critical mistake in evaluations to allow decision-makers or stakeholders to give weights to criteria – to specify trade-off relations – without specifying just how much change in the criteria is at stake (Keeney, 1992, p. 147). Accurate measurement of our separate concerns in a limited set of information-rich indicators can therefore help decision-makers better realise their trade-off preferences; since they are 'learning how to think'.[9]

2.6. MCCBA and ESS: synthesis

Having achieved this deeper understanding of the issues we may now come to a synthesis. The CBA and MCA methodologies have different evaluative strengths and weaknesses, but the MCCBA methodology combines the strengths of both techniques. First, the judgement criterion of an MCCBA criterion resembles that of CBA: it uses the costs and benefits of explicitly defined impact populations: 'regardless to whom they may accrue' within those populations. However, MCCBA often works with at least two impact populations or spatial levels. The first scale level stays close to the political or decision-making reality and thus is proximal to administrative borders.[10] The second scale level uses the global or the biggest spatial level on which impacts can be observed.[11] Analytically therefore, the impacts are divided between impacts in an 'own' region and 'other' or 'all' regions.

Second, in so far as measurement is concerned, MCCBA uses both CBA and MCA options. When choosing the best way of measurement, MCCBA uses the theoretical notion of *consensus based aggregation*, which implies that the procedure to measure impacts should obtain broad consensus among those involved in the evaluation concerning: (1) the minimal relevance of a criterion/effect measured and (2) the type of measurement, which should be both understandable and important to measure it that way (Sijtsma, 2006). In practice, the use of monetised measurement is confined to more market-economic and efficiency-related impacts and that, in addition to these monetised impacts, only a limited number of standardised impact measurement criteria are included.

Third, MCCBA, like CBA and MCA (but unlike the ESS framework as presented in MEA) is endpoint-oriented. If we examine the ESS framework with an endpoint view, we see that it is unusual that biodiversity concerns, or nature's well-being concerns in general, are not directly measured. This is odd for two reasons, first because the protection of nature is apparently the motive underlying the ESS framework. Second, the intrinsic value of nature, according to CBA analysts, is a separate concern, and is distinct from mere use values. The value of nature takes a significant position in the category of higher well-being, as we will see below. Surveys indicate that preferences for biodiversity conservation are not (only) services-based; concerns about nature and biodiversity are,

[7] Compare the tripartite Maslow-inspired scheme of Alderfer and comparable schemes (Rowan, 1998), and the composition of the Human Development Index (UNDP, 2009).

[8] Here, as elsewhere, there is a connection between higher and lower. Stable low-order distinctions form the basis for flexible high-order distinctions (Heylighen, 1992, p. 54).

[9] Here is where the MCCBA approach differs strongly from a CBA approach in which preferences are considered as given. They may be complex (or lexicographic) but they are nevertheless considered as given.

[10] This has a theoretical logic close to MCA, with its focus on the decision-maker.

[11] This logic is closely connected to CBA's a-spatial market stance.

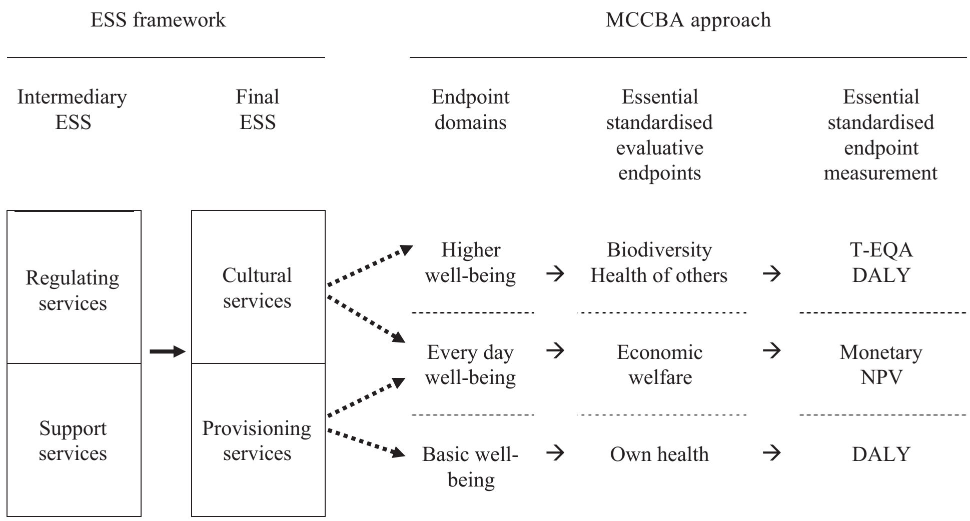

Fig. 2 – The relations between agro-ecosystem services and the standardised cardinal endpoint measurements of MCCBA.

first and foremost, moral preferences (The Gallup Organisation, 2007). In the MCCBA approach nature's well-being is measured separately.

Our combination of the ESS framework and MCCBA approach gives rise to the evaluative framework presented in Fig. 2.

The figure starts at the left with four groups of ecosystem services. Moving from left to right, the figure leads us to the *essential standardised endpoint measurement*. At the upper left the two groups of *intermediate ecosystem services (ESS)* are shown. In an evaluation, impacts on these intermediate services should preferably be assessed on the basis of their impact on the final ESS: cultural and provisioning services (Wallace, 2007). The procedure follows the CBA and MCA logic of avoiding double counting and redundancy of criteria. Nevertheless, a well-founded evaluation requires that the impacts be better structured, and this is shown as such in the right side of Fig. 2.

The final ESS impact on human well-being is divided in three endpoint well-being domains: basic well-being, every-day well-being and higher well-being. Within these encompassing well-being domains, MCCBA is focused on essential standardised evaluative endpoints and their possible measurements. As a standardised minimum, three endpoint measurements using cardinal/ratio measurements are presented: DALY (own health and health of others), monetary Net Present Value of a Cost-Benefit Analysis, and T-EQA (for biodiversity).

Basic health impacts can be measured in *Disability Adjusted Life Years* (DALYs), which is the World Health Organisation's standardised burden of disease measure (WHO, 2009).[12] With regard to hunger, lack of safety, changes in life expectancy, poor living conditions, and poor health, the DALY measurement is accurate, information-rich and easily (dis)aggregated. However, although the measurement of health in DALY is the same for 'own' health and others' (regions') health, the trade-off relations, i.e., the weight given to 'own' and 'others' DALYs, may of course differ substantially.

[12] Or Qalys, with less standardised weights, but several standardised ways to derive weights (Drummond et al., 2005).

The second endpoint, *economic welfare*, can be measured exclusively in monetary terms using a Cost-Benefit Analysis approach to valuation. For every day well-being a Cost-Benefit Analysis can capture a substantial amount of changes in well-being.

Cummins (1996) carried out an impressive inventory of indicators for well-being and classified them as the following: emotional well-being, community, intimacy, material well-being, productivity, safety domain, and health domain. Given this list, it may be disingenuous to argue for completeness of CBA to capture all impacts, which is certainly one reason for specifying higher and basic well-being domains. However, CBA clearly has much to offer in this intermediate 'everyday' level: on material well-being and productivity.

The third endpoint is higher well-being. This is part of the growth part of well-being. The higher the needs, the more the orientation of well-being of an individual or society (a system in Heylighen's terms) turns to the well-being beyond itself, and in various ways becomes more dependent on it (Koltko-Rivera, 2006). Fig. 2 shows two essential elements which may be recognised as recurring concerns in the debate on biodiversity, sustainability and ecosystem services (Sijtsma, 2006): the element of biodiversity (loss) and the element of the basic needs of others (often extremely poor people). For the latter, the health measurement of Disability/Quality Adjusted Life Years can be central. But how to measure biodiversity or, to mirror the human well-being term: nature's well-being? The next section will discuss a more accurate measurement of the concerns for biodiversity and explicate a new measurement concept for nature well-being: Threat weighted Ecological Quality Area (T-EQA).

3. T-EQAs in relation to agro-ecosystem services

We now turn to how the well-being of nature can be measured using Threat weighted Ecological Quality Area: T-EQA. Ecologists often use the term biodiversity to describe the well-being of nature. Biological diversity, or biodiversity, is the variety of life on earth, within species, between species and across ecosystems. The United Nations Convention on Biodiversity (CBD) uses a large set of indicators to monitor trends in biodiversity (EEA, 2010). The most commonly used indicators are the area of natural of semi-natural ecosystems and the numbers of species living within them. We propose here to combine both, that is, to measure the area of ecosystems as the natural unit (in hectares, or square kilometers), and then use species data to assess the quality of the area.

To that end we define Ecological Quality Area (EQA) as the basis of our nature value indicator. EQA is strongly related to the concepts developed by ten Brink (2000) and comparable to Strijker et al. (2000). The EQA combines two ecological aspects, namely the total area of an ecosystem or the combination of ecosystem[s] and ecological quality. Ecological quality is measured from 0 to 100%, depending on the quality level of an ecosystem compared to an ideal reference of the same type. The ecological quality of terrestrial systems can be based on the so-called *mean species abundance* (MSA) (ten Brink, 2000; MEA, 2005; ten Brink et al., 2002). The MSA provides an indication of the impacted state of an area relative to the pristine and un-impacted state of the ecosystem present in the area under consideration. From an agro-biodiversity perspective, the point of reference varies per ecosystem. Natural ecosystems have a references set at a natural state before human influence (as in the global MSA for biomes). More semi-natural or agricultural ecosystems have their own reference referring to natural ecosystems or systems with low-intensity farming. Modern high-intensive agriculture does not measure up to this 'ideal' reference point. If for example the Ecological Quality of an area with a specific ecosystem is 100%, then the health of the area's biodiversity is similar to the natural or low-affected state of this specific ecosystem. As such, from the perspective of individual ecosystems, the ecological quality can be regarded as an indicator of ecosystem intactness.[13]

However, the EQA indicator does not provide information on the relative preferences for various ecosystems. A hectare of intact semi-natural hayfield cannot be compared to a hectare of natural woodland. It is exactly this point which is so often at stake in evaluation because not every ecosystem type, or the species within them, is of similar societal concern, evidenced by the length of red lists of threatened species, the structure of many national and supra-national nature policies, and many (if not all) environmental impact assessments. As a general evaluation procedure it is therefore logical to apply weights to the different ecosystem types in order to reflect their significance in protecting biodiversity at a national or regional scale. To make this weight giving process uncontested, it should be based on objective ecological data on the degree of threat of the ecosystems or species under consideration. The larger the number or abundance of threatened species within an ecosystem type, the higher the loss of such ecosystem should be valued. To put it another way, the impacts of a project on biodiversity can be measured meaningfully when the 'local' intactness of ecosystem types is explicitly weighted with a value that indicates the influence of the EQA of a specific ecosystem type on the determination of the national or regional biodiversity. The weights for each specific ecosystem type should, to stress again, be based on objective systematic ecological data sets, such as the relative number of red list species within an ecosystem type.

The general procedure used to calculate a T-EQA score is to[14]:

(1) Determine the area of the different ecosystems – whether natural, semi-natural, agricultural, or urban – relevant to the project under consideration.
(2) Calculate the local intactness of the relevant ecosystems based on the presence or abundance of characteristic species relative to the number or abundance that would be present in an intact ecosystem. This yields a percentage score ranging from 0 to 100%. Rescale this ecological quality from 0 to 1 and multiply the scores for the different ecosystems by their area. This gives the EQA per ecosystem.
(3) Multiply the EQA of the ecosystems with a standardised weight factor indicating how much the ecosystem types contribute to the national or regional MSA. The relative number of red list species in an ecosystem can be used as a first proxy. The average weights of the eventual list of ecosystems on which the ecological evaluation data are based should be 1. Extremely threatened ecosystems should have the highest weight, while the most commonly occurring ecosystem with common species should have the lowest weight. The multiplication factor between the highest and lowest weight is what defines the Threat weight at a given spatial scale.[15]

This T-EQA measurement combines abundant ecological information in only one aggregated (cardinal) measure (Sijtsma et al., 2011). From an evaluation point of view, the T-EQA indicator is intellectually sound as well as intuitive in its measurement of nature impacts. It starts with the area of ecosystems (A); the bigger (or the more natural) an area is negatively (positively) affected by a project, the worse (better) the nature score of this project will be. Loss of high quality (EQ) hectares, e.g., forests with myriad forest species, is more costly than the loss of hectares with scant ecological forest quality. Furthermore, the more threatened (T) species are in the negatively (positively) impacted ecosystems, the worse is their loss (the greater the gain) in terms of T-EQA. All these

[13] For a worldwide application, the use of a historic reference is easiest due to data availability. One may also use other references than the historical pristine state. In targeting nature policies, use is often made of well-developed ecosystems elsewhere, in undisturbed or less disturbed contemporary conditions.

[14] See Sijtsma et al. (2009) for a detailed explanation in Dutch. See Sijtsma et al. (2011) for a re-evaluation with T-EQA of three Dutch CBA studies.

[15] Note that this weighing differs per spatial level. In the Netherlands for terrestrial systems we are currently working with a weight factor between the highest and lowest of 24: the weight given for "Threat" ranges from 0.1 (highly common ecosystem) to 2.4 (highly threatened ecosystem), around the median of 1.0 (Sijtsma et al., 2009, 2011).

elements are often measured separately in environmental evaluations but not in one measurement, and not using standardised cardinal/ratio measurement.

4. Discussion and research agenda

In our view, the MCCBA approach can be defined as an integrated and hybrid methodology: integrated because it is characterised by the use of both ecological, health and economic data, and hybrid because it is characterised by a multi-method use of alternative valuation methodologies that go beyond monetary measures alone. In the remainder of this paper we will discuss essential characteristics of the approach and the research agenda that it engenders.

4.1. *Testing T-EQA use and understanding*

To measure nature well-being we proposed T-EQA. If we examine T-EQA from a more technical measurement perspective within MCCBA, its consensus-based quality has to be addressed, i.e., are the expert weights regarding the degree of threat of ecosystems indeed uncontestable? And is the ecological quality measurement easy to understand by different stakeholders?

We have shown above that measuring nature well-being is a criterion with enough minimum relevance, but is the proposed measurement understandable, and is it widely seen as important to measure it in the way proposed? These issues should be tested in future research to ascertain how much its use helps decision-makers and stakeholders. This research is now underway in the Netherlands. From limited Dutch experience thus far, it is clear that T-EQA is greeted with substantial enthusiasm by both analysts and decision-makers (Wessels et al., 2011), but to CBA analysts its relation to welfare measurement is not always clear (De Blaeij and Verburg, 2011). Moreover, EQAs have proven to be more easily calculated consensus-based assessments than T-EQAs, and that further experience and debate are needed to establish stable threat weighing measurements for water and international biodiversity.

4.2. *Empirical measurements and completeness*

There is not an exhaustive set of standardised cardinal/ration measurements to account for every aspect of the three domains of well-being, but essential measurements have been developed within these domains. In the present paper we focus on three essential measurements: health in DALY, economic welfare in monetary NPV and nature's well-being or ecological quality in T-EQA, which we think are fundamental to ESS-related evaluations, as illustrated in Table 1. Table 3a depicts the impact of using MCCBA for project and policy evaluations that involve ESS. Instead of leading to a single score outcome for all policy options, Table 3a shows that the MCCBA approach generates separate scores in different well-being domains. Establishing trade-off relations, i.e., weights between these domains is not easy; it is far easier within one domain. The standardised impacts proposed in Table 3a are not complete, but they are essential. Each should always be measured[16] as a minimum. For the interpretation of impacts, a standardised and cardinal/ratio measurement is essential since it eases comparison of the size of impacts *across* projects and policies. Lists of services from the ESS framework as in Table 1 should be assessed as to their impact on the three endpoint concerns. Obviously, this may still present a substantial quantification research agenda however, ample non-contested reference numbers exist at different geographical levels.

To illustrate the relative ease of quantification, we present the hypothetical, though not farfetched, numbers in Table 3b. The table depicts 'state' indicators for ecological quality, health (i.e., burden of disease) and economic well-being, for three different types of countries and globally. These do not represent the change measurements that are pivotal to an evaluation; but rather these reference numbers show that for ecology, health and economic well-being, measureable information is available for all types of countries.

The MCCBA approach has as its focus cardinal/ratio indicators that can incorporate consensus-based weights in order to 'add-up' information within the value domains. At present, we do not have international weights for the Threat weights of the T-EQA; this lack should be further developed in research. The good news is that, at the global level, it is not actually the lack of data which complicates the setting of the weights; the problem here is to figure out how to test the weight factor in evaluation cases and also find a way to standardise at the global level.

Additional criteria covering impacts not measured by these essentials may be added to individual evaluations, crucially by following the proper rules of non-double counting and non-redundancy. Further research is needed as to the extent of the different dose–effect relations that can be established in different types of evaluations.

4.3. *Beyond the monetary valuation battleground*

What then does the combination of the ESS framework and the MCCBA approach bring to the evaluation of (agricultural) policies and management? As we have seen, using CBA leads to difficult monetisation efforts, while using MCA leads to many ordinal (+ or −) scores, and higher order weight assigning problems. MCCBA avoids monetisation where it is difficult and does not use weights for higher order value judgments between, for instance, health and economic well-being or between biodiversity and health. It only uses weights within these endpoint domains. Therefore quantification, which is so essential to decision support, can be easier, more standardised and more transparent. The MCCBA standpoint of specifying the impacts consistently for different regions is essential to MCCBA quantification.

The scores in the three domains are *not* added up. They can be interpreted in an objective way using ratio analysis (like cost-effectiveness analysis), which may help decision-making (this is a formal stage in the MCCBA approach; Sijtsma, 2006). Still, the bottom line is that MCCBA avoids the – commonly disputed – highest order weights, not because

[16] Provided of course the project or policy alternatives differ on the scores.

Table 3a – The basic decision-making scheme (from Table 1) and the impact of the use of MCCBA as an evaluation method.

		Policy or project options considered by the decision-maker:		
		A	**…**	**X**
Impacts evaluated by the decision-maker:	**1**	a1	…	x1
	2	a2	…	x2
	…	…	…	…
	ESS 30+	aess30+	…	xess30+

↓ ↓ ↓

MCCBA evaluation method

↓ ↓ ↓

Well-being domains:	Key indicators:			
Basic well-being	'Own' basic health (in $DALY_{own}$)	$aDALY_{own}$	…	$xDALY_{own}$
Every day well-being	Net monetary costs and benefits (in NPV)	aNPV	…	xNPV
Higher well-being	Nature well-being (in T-EQA) Health of others (in $DALY_{others}$)	aT-EQA $aDALY_{other}$	…	xT-EQA $xDALY_{other}$

Table 3b – Hypothetical though empirically inspired reference numbers quantifying the state of ecological quality, health (i.e., lack of health: the burden of disease) and economic welfare (for three different income levels of countries and the world).

	A low income country	A middle income country	A high income country	World
Reference values for key indicators				
DALY/1000 inhabitants	500	300	150	237
EQA/ha	0.75	0.65	0.40	0.72
GDP/capita	500	4000	18,000	9800

Source estimates: DALY from WHO (2009); EQA from NEAA (2008); GDP at www.indexmundi.com/.

it is inconvenient or theoretically non-optimal for decision support to provide the weights, but because in practice we see that they are difficult to provide in a consensus-based manner. MCCBA does, however, help interpret evaluation outcomes due to its standardisation and cardinal/ratio measurement; which will enable and enhance learning and comparison options. As we have emphasised earlier, on the subject of higher order trade-offs and higher well-being, we do not really know our preferences. We are learning how to think about them. The widely discussed distributional impacts in CBA – ethical concerns – are commonly seen within CBA theory and practice as best treated descriptively without fixed trade-off weights. Ethical concerns are an obvious example of higher well-being concerns, about which we are continuously 'learning how to think'. Moving up in the hierarchy of needs we tend to have increasing concern for the basic health and security of people other than ourselves. In line with this notion, the poor of the world suffering from famine and malnutrition increasingly depend on NGO actions in wealthy regions of the world (Watts, 2000).

The same holds for the issue of *intrinsic nature values*, or the closely related *non-use existence value* as it is labelled in the Total Economic Value concept (van der Heide et al., 2010; Sukhdev, 2010). Unlike the utilitarian ESS standpoint, evidence shows that preferences for biodiversity conservation are not purely service-based; many people have indicated that these preferences are primarily moral preferences (The Gallup Organisation, 2007). Strong moral motivations are typically higher needs, that is, growth needs (Heylighen, 1992). Maslow proved empirically that appreciation of nature is a key issue in higher needs-dominated people. The non-use appreciation of nature becomes increasingly important after the more functional services which nature has delivered to people are satisfied.[17]

The tripartite division in well-being may therefore help to define the proper domain where CBA-related monetary measurement can be fruitfully applied, that is, in intermediate, everyday well-being. Market values of quantities and price are extremely rich in information, be they wheat prices, land prices, housing prices, wage rates, etc. To measure changes in well-being using these prices and quantities, and account for the myriad trade-off relations within a CBA framework, is a highly useful and practical action. However, the more impacts affect our own basic well-being (valuing dominant and urgent needs as safety, hunger or lives) or the more they impinge on higher well-being (valuing non-urgent needs as ethics, aesthetics, intrinsic nature), the less useful will be the CBA-related monetary measurement of costs and benefits.

Further research will have to prove whether our analysis set out here fosters a more consensus-based approach, and if it actually helps decision-makers reconnect with nature without returning to the same old battleground that monetary valuation of ecosystem services has become.

[17] This 'Maslow logic' thus makes it easy to understand the dual-preference structure, the consumer-citizen split much discussed in the sustainability literature (Sijtsma, 2006).

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