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A critical review of reductionist approaches for assessing the progress towards sustainability

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Abstract

The increasing prominence of Sustainable Development as a policy objective has initiated a debate on appropriate frameworks and tools that will both provide guidance for a shift towards sustainability as well as a measure, preferably quantitative, of that shift. Sustainability assessment has thus the challenging task of capturing, addressing and suggesting solutions for a diverse set of issues that affect stakeholders with different values and span over different spatial and temporal scales. However sustainability assessment is still not a mature framework in the sense that Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA) are. This paper aims to provide suggestions for improving the sustainability evaluation part of a sustainability assessment. In particular it will provide a comprehensive review of different sustainability evaluation tools (from a reductionist perspective) as well as the feasibility of incorporating them within a sustainability assessment framework. Reviewed tools include monetary tools, biophysical models and sustainability indicators/composite indices that have been developed within different disciplines such as economics, statistics, ecology, engineering and town planning.

Keywords: Sustainability assessment; Reductionism; Monetary tools; Biophysical models; Composite sustainability index

Contents

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1. Introduction

The concept of Sustainable Development first appeared in the 1970s and early 1980s (IUCN, 1980) but only came into prominence following the 1987 World Commission on Environment and Development report where it was defined as "…development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (WCED, 1987). Several definitions have been proposed since then (refer to (Pezzoli, 1997) among others) but after almost 20 years of debate there seems to be a consensus that sustainability assessments ought to:

- integrate economic, environmental, social and increasingly institutional issues as well as to consider their interdependencies;
- consider the consequences of present actions well into the future;
- acknowledge the existence of uncertainties concerning the result of our present actions and act with a precautionary bias;
- engage the public;
- include equity considerations (intragenerational and intergenerational).

For the purpose of our paper a holistic sustainability assessment is considered one that is able to consider all these points simultaneously.

According to Raffensperger and Tickner (1999) the precautionary principle advocates that when an activity raises concerns or threats of harm to human health and the environment then precautionary measures should be taken even if some cause-and-effect relationships are not fully established scientifically. The precautionary principle is well rooted in the concepts of risk prevention and the ethical responsibility to prevent irreversible negative impacts.

If sustainability is perceived to be the ultimate planning goal then according to Briassoulis (1999) the question that has to be tackled by planners is:

"Given area X, what development path can be pursued over the next Y years (and beyond perhaps) so that the area's resources are used efficiently, its environment is protected and its economic welfare is promoted in a socially just way?"

However given the complexity of the systems that are usually the focus of sustainability assessments, e.g. regions, cities etc, and the multitude of the social actors involved, planning for and assessing the progress towards sustainability inevitably becomes a complicated task. In our view sustainability assessments have so far over relied on reductionist methodologies and tools. Munda (2006) comments on the characteristics of the reductionist paradigm for building descriptive models such as the use of a single measurable indicator (e.g. GDP per capita), a single dimension (e.g. economic dimension, in the triple bottom line sense), a single scale of analysis, a single objective (e.g. maximization of economic efficiency) and a single time horizon. Another distinctive characteristic of the reductionist paradigm is the tendency of quantifying and aggregating the different sustainability dimensions with a single unit of measurement. As it is obvious from our definitions, we do not suggest that a reductionist approach is the opposite of a holistic approach as we believe that they refer to different things. The former is relevant to the approach adopted towards understanding and describing a system while the latter is referring to the set of considerations that in our opinion have to be addressed by the analyst/stakeholders during the assessment stage in order to better inform decision making.

Reductionism, as discussed in this paper, despite the criticisms it has faced (refer to Section 6 for an overview) is still the dominant paradigm for sustainability assessments. According to Costanza (2000), the main advantage of reductionism stems from its "user friendliness" given its tendency to reduce the plethora of the environmental impacts of human activity to a limited set of numbers in order to integrate economic and environmental concerns in decision making. Such an approach thus seems to be compatible with the recurring appeal for "keeping it simple" expressed by stakeholders and policy makers. However there seems to be an abuse of reductionist tools on the part of the analysts in the sense that they are being indiscriminately used to supposedly measure the progress towards sustainability. In our opinion the different tools discussed in this paper illuminate different aspects related to the progress towards sustainability. In fact most of the methodologies and tools discussed in this paper were not designed for sustainability assessment per se and very few of them, if any, seem to be so flexible as to be able to assess the progress towards sustainability in a holistic manner. In our opinion reductionist methodologies and tools can produce a wealth of information and provide significant insights to the Sustainable Development debate. Furthermore it is possible to incorporate them in integrated assessment frameworks in order to better guide sustainability planning and evaluate policy proposals (refer to Section 5). Thus the aim of this paper is to provide a clear and concise picture of the scope, methodologies and assumptions of certain reductionist tools as well as their potential for assessing progress towards sustainability. The different current proposals are mainly assessed alongside our criteria for holism mentioned earlier and some of the points suggested by Lee (2006). Lee's points are mainly operational (in some cases overlap with our criteria for holism) and are concerned with the ability of an assessment methodology to:

- consider economic, environmental and social issues;
- predict future conditions under different scenarios;
- compare the likely outcomes of different action;
- communicate the results in an appropriate form to the stakeholders.

The approaches considered in this paper fall into three broad categories: monetary tools, biophysical models and sustainability indicators/composite indices. Their methodologies follow similar procedures relying on initial quantification and subsequent aggregation of the diverse sustainability issues using common denominators (e.g. money, energy etc).

Economic theory and monetary tools were probably the first proposals for assessing the progress towards sustainability (Pearce et al., 1989; Pearce, 1993a; Pezzey and Toman, 2002) but stemming from the inadequate expression of social and environmental sustainability issues in monetary terms the formulation and adoption of other measures was later proposed (Alberti, 1996; Ecological Economics, 1999; Munasighe and Shearer, 1995).

For the remainder of this paper and for reasons of conciseness the term tool will refer to an evaluation or aggregation methodology while the term metric will refer to the common currency/denominator that each tool utilises (e.g. Cost Benefit Analysis is the tool while money is its metric).

2. Monetary tools

Concepts and tools deriving from the economic literature and especially from the strands of welfare, development, environmental and more recently ecological economics have been utilised from the onset of the Sustainable Development debate to explain both its theoretical basis and to inform policy recommendations. The relevant economic literature is overwhelming and this paper does not attempt to provide a comprehensive review of it but it will offer insights into the mechanisms and assumptions of three tools that, in our view, have potential for sustainability assessment; the Contingent Valuation Method (CVM), Cost Benefit Analysis (CBA) and the Index of Sustainable Economic Welfare (ISEW).

Neoclassical economists are also responsible for the designation of two central concepts related to Sustainable Development; strong and weak sustainability. These concepts stem from the tendency of economists to distinguish between the different forms of capital namely man-made capital, human capital, social capital and natural capital. According to Neumayer (2003), to most economists the definition of Sustainable Development as "…development that does not decrease the capacity to provide non-declining per capita utility for infinity" would seem acceptable. Supporters of weak sustainability believe that this non-declining utility can be achieved through total substitution between the different forms of capital (Solow-Hartwick rule) e.g. (Hartwick, 1977, 1978a,b, 1992; Solow, 1974, 1986, 1993). Supporters of strong sustainability, on the other hand, generally consider natural capital as non substitutable with other forms of capital (Daly, 1992, 1996). A commonly evoked argument by proponents of strong sustainability is the necessity to impose limits on decisions that have a detrimental effect on the legacy of future generations by irreversibly decreasing the capacity of the environment, or of society, to provide important functions. A third approach resting between weak and strong sustainability but somewhat closer to strong sustainability, regards natural capital substitutable only to a degree with the other forms of capital (Pearce, 1997). This limitation is imposed by physical and technological constraints (Ayres, 2007), leading to the concept of Critical Natural Capital (Noel and O'Connor, 1998). In our view the concepts of weak and strong sustainability have gone beyond the realm of economics (i.e. substitution between the different forms of capital) to indicate the presence or absence of trade-offs between different sustainability issues and they will be used in this sense for the remainder of the paper.

The first question that comes to mind is why somebody would use money as a metric of Sustainable Development. Pearce et al. (1989), argues that money can play a significant role in assessing the progress towards sustainability since money is a good indicator not only of the public's Willingness To Pay (WTP) but also of the intensity of that preference. Furthermore, several tools allowing the comparison of monetary values have been developed rendering the comparison between monetised quantities relatively easy and straightforward (ibid). A final commonly evoked argument is that monetary values tend to be more easily understood by non-experts and relevant stakeholders.

On the other hand, monetary analysis of sustainability has been rejected mainly on ethical grounds and the fact that it may be devoid of common sense in some cases with Heinzerling and Ackerman (2002) providing illustrative examples. Another relevant criticism is its over-reliance on subjective valuations (Willingness To Pay or Willingness To Accept) thus viewing sustainable development through a rather anthropocentric prism. An immediate implication of this is that certain monetary valuations would be only valid if the evaluators had perfect information on the functioning and interrelations of the system under study. However, this is not usually the case due principally to scientific uncertainty and ignorance in many relevant fields of the environmental and social sciences.

Several valuation techniques are being used for the monetisation of sustainability issues. Of the different evaluation techniques included in Table 1, the Contingent Valuation Method (CVM) has emerged as the most widely used for the quantification of sustainability issues that have usually been considered in the past as intangibles. For this reason CVM is the only monetary valuation technique reviewed in depth in this paper. The interested reader is referred to Hanley and Spash, (1993), Pearce et al. (1989), Pearce (1993b) and Tietenberg (2003) for further details on other monetary valuation techniques. Monetised quantities are subsequently fed into aggregation tools. Aggregation tools that are reviewed in this paper are the Cost Benefit Analysis (CBA) and the Index of Sustainable Economic Welfare (ISEW).

Table 1
Economic valuation methods for quantifying sustainability issues (Tietenberg, 2003) adapted from Freeman (1993)

Methods	Observed behaviour	Hypothetical
Direct	Market Price Simulated Markets	Contingent Valuation
Indirect	Travel Cost Method Hedonic Property Values Hedonic Wage Values Avoidance Expenditures	Contingent Ranking

2.1. Contingent Valuation Method (CVM)

CVM was conceived 60 years ago by Ciriacy-Wantrup (1947) and entered the mainstream following the formulation of the concepts of non-use values in the 1960s (Venkatachalam, 2004). Its growing popularity and the controversy that followed its use in the aftermath of the Exxon Valdez oil spill was perhaps the catalyst that influenced the US National Oceanic and Atmospheric Administration (NOAA) to assemble an expert panel, which included the Nobel Prize laureates in Economics K. Arrow and R. Solow amongst others, to comment on the technique (Arrow et al., 1993).

In a nutshell, CVM analysts aim to capture the stated preference of the public regarding a particular "good/service" by measuring its Willingness To Pay – WTP for that given "good/service" or its Willingness To Accept – WTA compensation for the loss of that "good/service". The term "good/service" is included in quotation marks because these goods/services are not usually traded in real markets. In order to assign such monetary values a hypothetical market trading them is created and the monetary value is elicited from the public through direct questions administered in the form of surveys, questionnaires or interviews.

As a result of its methodological procedure the CVM is theoretically capable of evaluating a plethora of sustainability issues that cannot be captured by other monetary valuation techniques. Some of the issues that have been quantified in recent years include urban green spaces (Tyrvainen, 2001; Jim and Chen, 2006), urban river quality (Bateman et al., 2005), road noise (Fosgerau and Bjorner, 2006), water quality (Atkins and Burdon, 2005), biodiversity (Christie et al., 2005), health (Habbani et al., 2006; Johannesson et al., 1996; Mataria et al., 2006), culture

(Dutta et al., 2007; Thompson et al., 2002) and education (Stair et al., 2005).

Venkatachalam (2004), in his comprehensive literature review reports that the main criticisms revolve around two concerns: validity and reliability. Perhaps the key concern over its validity is the observed discrepancies between WTP and WTA values for the same good with the WTA values always being greater than the WTP values (Brookshire and Coursey, 1987; Coursey et al., 1987; Hanemann, 1991). Nevertheless, according to CVM's theoretical foundations, both WTP and WTA can be used to obtain monetised preferences (Venkatachalam, 2004). Arrow et al. (1993) proposed the use of the WTP in preference to WTA in CVM studies by avoiding WTA scenarios but clearly in some cases this is not possible. Other issues affecting validity include:

- embedding of a good/service (i.e. differences in WTP for a good/service when it is valuated on its own or as part of another good/service);
- sequence of the questions in the survey;
- provision of adequate information;
- the elicitation technique;
- hypothetical bias i.e. difference between the WTP indicated by the respondents and what they actually pay when they are asked to contribute;
- the strategic bias (e.g. free riding by stating lower WTP believing that other respondents would be willing to pay more).

The level of information provided to the respondents of the survey is considered very important and although several studies have been conducted to determine the optimal level their findings are ambiguous. Blomquist and Whitehead (1998) believe that adequate information is important to reduce the uncertainties of the valuation exercise while Harris et al. (1989) suggested that excessive information might have a detrimental effect by confusing the respondents.

A major concern related to the reliability of CVM surveys is whether and to what extent it is possible to generalise the findings of a CVM study. If, in theory, it was possible to apply the elicited values from a CVM study to predict the stated preference in other contextually similar case studies then significant time, effort and money could be saved. However, studies have shown that, although, this "benefit transfer" is possible under certain circumstances (Bateman et al., 2002) it is not generally very dependable (Kirchhoff et al., 1997).

Most proponents of the CVM believe that the aforementioned problems stem from the survey design and could be avoided by better structured surveys. A number of different relevant suggestions found in the literature (Arrow et al., 1993; Bateman and Turner, 1993; Cummings et al., 1986) have been collected by Venkatachalam (2004). Additionally, MacMillan et al. (2002, 2006) suggested that one shot surveys might be inadequate for the accurate valuation of non familiar environmental goods/services and proposed the "market stall" approach. Valuation in this case is not one shot but is performed after discussion of the issue and exchange of information by small groups of evaluators. However, the incorporation of all the suggestions found in the literature could dramatically increase the cost of a CVM study raising doubt on its viability for small scale projects.

Heinzerling and Ackerman (2002) offer a third set of criticisms focusing more on the underlying philosophy of the CVM rather than its methodological aspects. The first point reflects the fact that respondents in CVM surveys are asked to give their preferences as consumers rather than as citizens. Answers from persons assuming these two different roles are strikingly different as exemplified by Sagoff (1988). Their second point concerns the tendency to ask respondents to give their preferences as individuals without considering the effects on others. For example parents' WTP or WTA will most probably be different when they consider, in their valuation, only themselves or themselves and their children.

2.2. Cost Benefit Analysis (CBA)

Cost Benefit Analysis (CBA) is one of the most widely used project and policy appraisal tools that has gained prominence over the past decades. Despite the fact that CBA as a generic tool can be adapted to utilise metrics different from money, e.g. employment (Taylor, 2001), the fact remains that very few, if any, sustainability assessments have attempted that so far.

CBAs are performed during the early planning stages of a project/policy in order to determine its feasibility and have usually been used to answer two main sets of questions: whether a project X be undertaken or which of a series of projects X, Y, Z, etc should be undertaken.

Assuming that there will always be people that gain and lose from a given project/policy, these questions are answered through a comparison of the sum of the anticipated benefits of the proposals quantified as the gainer's Willingness To Pay for these benefits and the sum of the anticipated costs quantified as the loser's Willingness To Accept these costs. The project/policy that results in the greatest welfare improvement for the affected social actors is the most acceptable.

Layard and Glaister (1994) proposed the following generic procedure for a CBA:

- valuation of the costs and benefits for each year of the project/policy;
- discounting of the costs and benefits in future years to make them commensurate with present costs and benefits;
- calculation of a Net Present Value (NPV) by aggregating and comparing costs and benefits over the whole life of the project/policy.

CBAs usually utilise one or more of the valuation tools included in Table 1 for the monetisation of the applicable costs and benefits. However, monetisation can insert significant uncertainties jeopardising the quality of a CBA such as the discrepancies between WTP and WTA mentioned in the previous section.

In a setting with several issues expected to influence the progress towards sustainability and a number of social actors affected CBA can end up being a rather complicated procedure.

Immediately, three significant problems arise when dealing with the costs/benefits: their valuation, their discounting and their aggregation/comparison. Of the three, valuation has been discussed in the previous section. Discounting and aggregation/comparison will be discussed below. We note here that current discounting and aggregation practices have been criticised as contrary to one particular aspect of Sustainable Development; the need for intragenerational and intergenerational equity.

Discounting is an important albeit controversial part of the CBA which is performed in order to compare future costs/benefits with present costs/benefits, essentially representing the trade-off between the enjoyment of present and future benefits (Howarth, 1996). The greater the discount rate adopted the greater the devaluation of distant future impacts (whether costs or benefits). As a result future impacts count for little in projects/policies with a long time horizon especially those expected to span over different generations. This can be perceived as contrary to the interests of future generations, leading to a non equitable distribution of costs and benefit through time by forcing future generations to bear a disproportionate cost (Pearce et al., 1989). In order to overcome this situation it has been suggested that low discount rates should be adopted for projects that will greatly affect future generations while in some cases such as mortality and species extinction adoption of a zero discount rate is justifiable (Daly and Cobb, 1989). Saez and Requena (2007), report that certain economists take this rationale even further by suggesting that the only discount rates perfectly aligned with the appeal for intergenerational equity are discount rates of zero. However, choosing artificially low discount rates as the sole measure of safeguarding the interests of future generations does not necessarily guarantee intergenerational equity, (Howarth, 1996), as it might increase the possibility of ignoring other policy issues. It also lacks theoretical foundations especially when it is inconsistent with the prevailing interest rates (Pearce et al., 1989). For example if net future cash flows are positive low discount rates might trigger more present investments in the public sector resulting in an increase on the demand of environmental resources and thus accelerating their depletion and the accompanying environmental degradation (Bowers, 1997; Pearce et al., 1989). Rabl (1996) illustrated the potential of facing criticism from future generations for not safeguarding their interests by not discounting properly. What is more important though is that generally speaking questions on intergenerational equity must be judged according to ethical values, something that is conceptually distinct from the concept of economic efficiency on which CBA is based (Howarth, 1996) quoting Howarth and Norgaard (1992, 1995). Padilla (2002), traces this apparent lack of consideration for future generations not on the absence of an equity criterion but on the rather arbitrary extension of time preferences of present generations beyond their existence.

Another important problem in CBA studies arises from the criteria being used for the specification of welfare improvement resulting from a project/policy. Layard and Glaister (1994) note that if the Pareto criterion (i.e. all social actors affected by the project/policy gain and nobody loses by ensuring adequate compensation of the losers by the winners) were used in a CBA setting it is likely that no project will ever be undertaken since it would be almost impossible to compensate every social actor negatively affected by the given project/policy. For this reason most CBA analysts have utilised the Kaldor–Hicks criterion, essentially a necessary condition of the Pareto criterion (Stavins et al., 2003), despite the lack of any ethical justification (Layard and Glaister, 1994). The Kaldor–Hicks criterion states that a project/policy could be undertaken if the size of the benefits is such that the gainers could compensate the losers in theory, though the compensation would not have to be actually carried out (Brent, 1996). So one can deduce that the Pareto criterion seeks to safeguard the welfare of every social actor affected while the Kaldor–Hicks criterion safeguards the welfare of the society thus allowing a project/policy to be undertaken even if some social actors lose, and are not compensated, provided that society gains as a whole. One of the hidden assumptions of the Kaldor–Hicks criterion is that it treats the marginal utility of a unit of additional income as the same for all social actors (Munda, 1996), something that is rarely the case as has been pointed out in the past e.g. (Bernoulli, 1954) cited by Quiggin (1997), (Layard, 2005). A way to overcome this is by utilising distributional weights to allow for differences in satisfaction from additional income but as Farrow (1998) points out such weights can be considered arbitrary and can be contested whilst at the same time further complicating the CBA procedure. Munda (1996), expresses a much more important reservation for using weights in CBA stemming from the linear aggregation approach of CBA and that is that the weights lose the concept of importance and adopt a meaning of trade-off ratio. According to Zerbe et al. (2006), the current version of the Kaldor–Hicks criterion is characterised, amongst others, by the fact that equity considerations are disregarded. Similar findings regarding the inadequacy of the Kaldor–Hicks criterion, especially in respect to environmental equity and sustainability, are noted by Farrow (1998). Finally, the applicability of the Kaldor–Hicks criterion has been contested for intergenerational settings (Azar, 2000).

2.3. Index of Sustainable Economic Welfare (ISEW)

The Index of Sustainable Economic Welfare (ISEW) (Daly and Cobb, 1989), the Genuine Progress Indicator (GPI) (Redefining Progress, 1995) and the Sustainable Net Benefit Index (SNBI) (Lawn and Sanders, 1999) are three closely related sustainability accounting frameworks that build on earlier attempts such as those by Nordhaus and Tobin (1972) and Zolotas (1981). All three were formulated in a way to better approximate the sustainable economic welfare of a given population and specifically to provide an alternative to other national account measures such the Gross Domestic Product (GDP) that were deemed to be inadequate for capturing human welfare. Even though these tools have been mainly computed at the national level e.g. (Clarke and Islam, 2005; Castañeda, 1999; Hamilton, 1999; Lawn and Sanders, 1999) a few recent studies have used them for sustainability assessments at smaller scales such as the regional and the urban scale (Costanza et al., 2004; Pulselli et al., 2006; Redefining Progress, 2004). Given their similarities and for reasons of conciseness the term ISEW will denote all three frameworks for the rest of the paper.

The first step of the ISEW's methodological approach is the weighting of the personal expenditures of the given population with an index of income inequality. Subsequently this figure is modified by adding and subtracting the monetary values of a predetermined set of factors that are deemed to affect the overall welfare of the population in a positive or negative manner. Such factors encompass a broad variety of environmental, economic and social issues such as the costs of consumer durables, environmental pollution, commuting, crime, unemployment and lost leisure time to name just a few. For a comprehensive list of the issues covered by each accounting framework the interested reader is referred to (Daly and Cobb, 1989; Redefining Progress, 1995; Lawn and Sanders, 1999). Increasing levels of ISEW over time imply increasing welfare and a positive progress towards sustainability and as a result measures that promote this increase should be preferred.

Dietz and Neumayer (2006), comment on the high likelihood of no two ISEW studies being similar as a result of the utilisation of different valuation techniques and the exclusion of certain items. This absence of consistency probably stems from the fact that the ISEW is a very data intensive framework and in some cases quantification of certain items is problematic or even impossible due to lack of good quality data. Less frequently, generic items have been added to the ISEW by certain analysts in order to highlight special characteristics of their case studies. Two such examples are the case of Thailand's ISEW (Clarke and Islam, 2005) that included generic defensive expenditures such as corruption and commercial sex work and the case of the province of Siena that included the cost of urbanisation (Pulselli et al., 2006).

Despite its inconsistency, the ISEW has gained some acceptance since its inception. Clearly, the most important reason is the fact that ISEW accounts for certain environmental and social issues, in addition to the usual economic issues, so it can be argued that it offers a better proxy for assessing welfare. For example, England (1998) and Stockhammer et al. (1997) argue that the ISEW is a great leap forward when compared to other national accounts frameworks such as the GDP but further modifications are essential.

However, ISEW has also faced significant criticism (Dietz and Neumayer 2006; Neumayer, 1999, 2000, 2003). The criticisms of the ISEW revolve around two main axes; its lack of theoretical validity and the tendency of some of its proponents to regard it as a validation of the "threshold hypothesis" proposed by the Chilean economist Max–Neef. The threshold hypothesis states that:

"…for every society there seems to be a period in which economic growth (as conventionally measured) brings about an improvement in the quality of life, but only up to a point – the threshold point- beyond which, if there is more economic growth, quality of life may begin to deteriorate." (Max-Neef, 1995).

Regarding this second set of criticisms (Dietz and Neumayer, 2006; Neumayer, 2000, 2003) showed that some of the effects of the threshold hypothesis, especially those concerned with the valuation of the depletion of non-renewable resources, the valuation of long-term environmental damage and the adjustment of consumption expenditures for income inequality seem to be reduced by using different valuation techniques than the ones initially proposed. This finding implies that ISEW results are sensitive to the choice of valuation methodologies for the different items of the index so significant attention must be paid to that choice in order to assure the quality of the results and not send misleading policy recommendations.

Lawn (2003), on the other hand, responding to the first set of criticisms proved the theoretical soundness of the three frameworks by tracing their origins to Fisher's concept of income and capital (Fisher, 1906). More specifically it was argued that Fisher's concept of income (i.e. economic welfare depends on the psychic enjoyment of life) is far superior to Hicks' view of income (Hicks, 1946) (i.e. economic welfare depends on the rate of production and consumption) when it comes to the assessment of a population's welfare. However, according to Dietz and Neumayer (2006), some of the methodological choices in the three indices can be seen as contrary to Fisher's view of income evoked by Lawn (2003). Another aspect of the ISEW that has been contested is the choice of the items included. It has been argued that some important items that might be expected to increase the welfare of a given population over time such as better quality consumption goods or life expectancy are excluded (Neumayer, 2003).

Perhaps the most important barrier particularly for the study of smaller scale systems (e.g. regional and urban systems) is the fact that generally data availability decreases with decreasing scale. This finding has been confirmed by the multiscale comparative analysis of Costanza et al. (2004).

2.4. Some comments

One of the major criticisms of sustainability assessments based on monetary analyses is the problematic valuation of the diverse sustainability issues, especially those that cannot be translated meaningfully into products/services traded in existing markets. Even though there are certain quite robust valuation techniques most of them fail to capture such issues in a non-controversial manner. CVM, the most commonly used such technique suffers from its highly subjective valuation procedure with various errors being attributed to a string of factors including the quality of the information provided to the public, strategic answers and the difficulty of generalising the findings of a CVM study. Additionally theoretical inconsistencies such as the discrepancies between WTA and WTP raise further doubts about its robustness.

CBA is rooted in the concept of economic efficiency and not of distributional equity and justice. Certain economists argue that issues of intergenerational equity cannot and should not be addressed by CBA alone (Goulder and Stavins, 2002). Taking this rationale a little bit further one can reason that notions of intergenerational equity cannot be addressed by explicit choices of an "optimum" discount rate (whether this rate is low or zero) but instead must be addressed by other means. One possible way is through the establishment of institutional frameworks and mechanisms that aim to safeguard the rights of future generations since future generations cannot represent themselves in contemporary markets and institutions (Padilla, 2002).

Other economists, however, believe that they must not only provide policy makers with information on economic efficiency but must also be ready to include in their analyses additional policy questions (Sumaila and Walters, 2005). Some of the proposals aligned with that view include:

- the development of two step discounting procedures that adopt a conventional discount rate for the first generation and the growth rate of the economy after that point (Rabl, 1996);
- the consideration of costs/benefits to the present stakeholders (at a standard rate) and to each future increment of stakeholders at that same standard rate but only after they enter the population (Sumaila and Walters, 2005);
- the simultaneous utilisation of a common discount rate for market goods and a lower discount rate (environmental discount rate) for non-market goods (Saez et al., 2007).

Whatever the case one must keep in mind Tol's (1999) argument that decisions on discount rates and methodologies are not trivial and must be guided both by empirical evidence and ethical judgement. Similarly, for intragenerational equity, alternatives such as an improved Kaldor–Hicks criterion that encompasses moral sentiments (Zerbe et al., 2006) or even the adoption of welfare criteria additional to the Kaldor–Hicks criterion have been proposed (Farrow, 1998). A rather different proposal has been articulated by Munda (2006). In particular it has been suggested that in certain cases, e.g. in CBAs dealing with cultural heritage, the compensation principle should be radically transformed to the precautionary principle.

Munda's (2006) previous comment hints that the mentality behind CBA is conceptually different from that of the precautionary principle. That seems to be quite true and there is a number of different explanations documented in the literature. The European Environmental Bureau states that a formal CBA is in contradiction with the precautionary principle since it assumes certainty where, by definition, certainty does not exist (EEB, 1999) while Kuntz-Duriseti (2004) states that the application of the precautionary principle to environmental risks reflects the concern that the traditional CBA is insufficient in cases of high scientific uncertainty. A methodological choice of CBA that seems incompatible with the precautionary principle is the equal weighting of costs and benefits. This equal weighing implies that a unit of loss is valued equally with a unit of gain which is the assumption of risk neutrality (Pearce, 1994). However, the precautionary principle is rooted in the concept of risk aversion rather than of risk neutrality, refer to the Introduction. A possible way to overcome this is by assigning high weights to relevant costs. However, some of the problems that arise by assigning weights in CBA have already been discussed in Section 2.2. Proposals to reconcile CBA with the precautionary principle include the evaluation of the effects of uncertainty (Kuntz-Duriseti, 2004), different weighing of costs/benefits (Pearce, 1994) and qualitative CBAs (van den Bergh, 2004). Similar conclusions can be drawn and for the ISEW.

Despite their differences in scope and methodologies, CBA and ISEW share a very important common feature. Both tools are based on the perspective of weak sustainability. CBA for example seeks to maximize the welfare of the different social actors by allowing tradeoffs between different monetary quantities. Munda (1996), traces the weak sustainability nature of CBA to the fact that since the "...concept of intensity of preference is used, complete compensability among attributes is allowed". Substitutability stems from the adoption of equal weighting and linear aggregation resulting in the weights to adopt a trade-off status. These tradeoffs between costs and benefits imply weak sustainability in the sense described earlier. Similar conclusions can be drawn for the ISEW where maximisation of the index over time would imply a progress towards a more sustainable society.

A final comment is on the diffusion of the ISEW among planners and policy makers. Despite its strengths there is no sign of the ISEW ever having been used outside academia for planning and policy making. There seem to be two main reasons for this. First of all, the ISEW has been developed for the assessment of a population's welfare at high levels of hierarchy with the city level as the smallest scale ever attained. This means that the ISEW seems capable of only capturing impacts on welfare resulting from large projects/policies as these kinds of projects are most likely to have a discernible effect on the welfare of the society. Secondly and most importantly, the ISEW does not seem to be able to quantify ex ante the effect of a major project/policy on the sustainable economic welfare of a population. In other words ISEW has not been used to answer questions such as "how much will project/policy X affect the ISEW" but it can theoretically assess ex-post the effect and monitor the impact of a major project/policy on the welfare of the population after its completion. Another possible use of ISEW is to act as a guide to other project/policy appraisal tools such as the CBA by indicating key areas that are bound to affect the welfare of a population both positively and negatively and that need to be quantified and included in their calculations. Additionally, it can be used to assess the welfare levels of a given population at the early stages of the planning process in order to identify priority areas that ought to be addressed or that should not be affected negatively by the project/policy in question.

3. Biophysical models

The shortcomings and limitations of the monetary approach was the major driver for the formulation of biophysical metrics and tools for assessing the progress towards sustainability. Three promising techniques that show potential for sustainability assessment at various scales are emergy, exergy and the ecological footprint.

3.1. Emergy

The term emergy was coined by D.M. Scienceman and H.T. Odum and has been defined as "…the available energy of one kind that has been previously used up, directly and indirectly, to make a service or a product and its unit is the emjoule (ej)" (Odum, 1988; Scienceman, 1987). Given the different abilities of each form of energy to produce work and the fact that different forms of energy are used in complex systems, emergy synthesis expresses all these different kinds in units of available solar energy that would be required in order to generate all inputs to the system. Thus, solar emergy can be defined as "…the available solar energy used up directly and indirectly to make a service or a product and its unit is the solar emjoule (sej)" (Odum, 1996). It should be noted here that emergy is not energy and thus is not conserved as such (Sciubba and Ulgiati, 2005). In recent years emergy synthesis is being increasingly used to quantify the environmental support required by human systems and eco-systems for all the inputs that are required by them while the total emergy consumed within the system is an indication of its total appropriation of environmental services (Ulgiati et al., 2006). As a result the emergy content of a product/service can be viewed as a measure of sustainability and/or pressure on the environment (Sciubba and Ulgiati, 2005). In a nutshell, emergy synthesis can be described as an accounting methodology that can assess the relationships between human systems such as the economy and their support environment. This is feasible because the work of both is expressed in equivalent terms (Campbell, 1998). At the centre of emergy synthesis lie the two assumptions that in every observable phenomenon there is energy transformation and that all these energy transformations can be accounted for as energy of one kind (solar emergy).

A key concept of the emergy theory is Solar Transformity that is essentially a measure of energy quality and is defined as "…the solar emergy required to make 1 J of a service or a product (it is measured in sej·J^{-1})" (Odum, 1996). Transformities are thus quality factors that highlight the amount of environmental work required for the production of a product/service.

Emergy synthesis is a "top-down" three stage process (Odum, 1996; Sciubba and Ulgiati, 2005); formulation of emergy diagrams, construction of emergy evaluation tables and calculation of emergy indices that allow for the evaluation of the system's performance. Emergy diagrams depict the different material/energy/monetary etc flows within a system and use a series of specific symbols to indicate the different flows, storages interactions, etc within the system (Odum, 1996). Their construction is a learning exercise and greatly assists the appreciation of the different issues affecting the sustainability of the system under study. The different emergy flows within the system are aggregated to provide a simplified picture of the metabolism of the system. Relevant aggregated flows are:

- Renewable resources: includes the free ecological services such as sunlight, rain, wind, waves, rivers, tides etc as well as renewable energy sources (hydroelectricity, Aeolian power etc), agricultural production, livestock and timber harvest;
- Non-renewable production: includes fossil fuels, metals, minerals and soils. Non-renewable production is divided into dispersed rural resources (such as soil) and concentrated resources;
- Imports/exports: includes fuels, goods/minerals and services.

Emergy indices summarise flows of emergy within the system under study and can provide a valuable decision-aiding tool (Ulgiati et al., 1995; Ulgiati and Brown, 1997, 1998, 2002). Some of the most commonly used indices include the:

- Emergy used per person: aggregates the different emergy flows used within the system and divides it with the population of the system. Generally speaking the higher the emergy used per person the higher the standard of living;
- Emergy Investment Ratio (EIR): the ratio of purchased emergy flows to a system divided by the free emergy inputs (indigenous). The lower this ratio the more the system relies on local ("free") indigenous inputs;
- Environmental Load Ratio (ELR): the ratio of the non-renewable emergy flows consumed within a system divided by the renewable emergy flows for that system. High Environmental Loading Ratios imply high environmental stresses;
- Empower density: the ratio of the emergy used within the system divided by the total area of the system. High empower densities are usually a characteristic of industrialised nations.

A fourth step that can further facilitate the communication of the results and the decision making process is the development of ternary diagrams which visually represent some of the indices mentioned above and can assist the comparison of different development paths (Almeida et al., 2006; Giannetti et al., 2006).

An interesting feature of emergy synthesis is its approach towards linking environmental production and its economic use. According to Odum (1996) real wealth derives from environmental resources measured by emergy while the buying power of money on average depends on how much real wealth (emergy) there is to buy. Thus, by dividing the emergy use of an economy with the money circulation within it one can find the emergy-to-money ratio that denotes the "real" buying power of money. The proportion of buying power due to an emergy flow is measured in emdollars (Em$), where the emdollar value of a flow or storage is defined by its emergy value divided by the emergy-to-money ratio for an economy for a given year (ibid). This concept is amongst the most controversial aspects of emergy theory and has evoked mixed reactions. This idea has also influenced other biophysical models such as parts of the Extended Exergy Analysis.

3.2. Exergy

Exergy (or available energy) is a thermodynamic property of a system that has been widely used by engineers to study and improve the efficiency of chemical and thermal processes. The roots of exergy analysis can be traced to the 19th Century and the pioneering work of S. Carnot and W. Gibbs while the term was coined in the mid 1950s by Z. Rant. Over the past years exergy has also gained popularity amongst certain engineers and ecologists for the study of complex human and ecological systems.

Exergy can be defined as the maximum work that can be extracted from a system when this system moves towards thermodynamic equilibrium with a reference state. Thus, the exergy of an energy form is an entropy free form of energy that can be perceived as a measure of its usefulness or quality or potential to cause change. Exergy, in contrast to energy, is not subject to a conservation law except for ideal or reversible processes but is consumed or destroyed due to irreversibilities. In fact, the concept of exergy derives from a combination of the first two Laws of Thermodynamics (Ertesvag, 2001) since it is a measure of both the quantity (1st Law) and the quality (2nd Law–Entropy Law) (Dincer, 2002) of different energy sources and is defined only after the selection of a reference environment (refer to Table 2 and (Valero, 2006) for the implications arising from the designation of a reference state). A detailed description of the thermodynamic foundations and the methodology adopted for calculating the exergetic content of substances as diverse as fuels, minerals, pollutants and industrial products is included in Kotas (1985) and Szargut et al. (1988).

Exergy accounting is a useful methodology that can provide insights into the metabolism of a system and the effect of the system on the environment using a common denominator while it can form an indispensable part of the "toolkit" needed to secure a sustainable future (Hammond, 2004). Dincer (2002), comments that exergy accounting can play a significant role for the provision of policy advice on energy planning and sustainable development as it:

- can address the impact of energy utilisation on the environment;
- is ideal for the design and analysis of energy systems as its methodology combines the conservation of mass and energy with the second law of thermodynamics;
- quantifies waste and energy losses so it can provide important information for more efficient resource use.

Furthermore, Bastianoni et al. (2005) state that the use of thermodynamic, and exergy in particular, indicators for sustainability assessment is justified as a result of Herman Daly's first principle of sustainability. According to Daly (1990) resources should not be used quicker than they can be regenerated. This implies that some metrics appropriate for material and energy balances, such as exergy, should be adopted as part of a comprehensive sustainability assessment. Central to exergy analysis is the concept of exergy efficiency which according to Ulgiati et al. (2006) has a direct relationship with the market logic. Rosen and Dincer (2001) discussing the links between exergy efficiency and

Table 2
Attractive features and criticisms of the three biophysical approaches

Approach	Attractive Features	Criticisms
Emergy	• Considers free ecological processes • Able to compare disparate materials/energy sources • Broad scope that can address economic, environmental and social issues	• Ignores human preference • Uncertainty of utilised Transformity values • Some allocation decisions • Maximum Empower Principle
Exergy	• Based on a well defined methodology that is universally accepted • Obtained from laws of thermodynamics that allow precise measurement • Identifies the presence of irreversibillities in the various steps of a production chain so in a sense it can locate the "weakest link"	• Ignores human preference • Requires the definition of a reference state. This introduces a sense of relativity as the result of the analysis depends on the choice of the reference state. • Cannot identify the causes of the irreversibilities.
Ecological Footprint	• Focuses on consumption and gives information on the demand of human societies on natural ecosystem support • Accounts for population size and resource consumption, both aspects of environmental load • Simple, flexible and visual tool that can be communicated easily to the non-expert. • Recently underwent a standardisation process (Global Footprint Network, 2006)	• Weak analytical tool • Calculation of energy component (CO_2 land) • Cannot consider pollution and important environmental impacts other than those that can be translated into loss of bioproductive land • Inability of quantifying resource depletion that cannot be translated to a bioproductive area (e.g. minerals)

sustainability mentioned that an increase in exergy efficiency implies a decrease of environmental impact (exergy conversion with fewer losses) and an increase in sustainability (process approaches reversibility).

In contrast to emergy synthesis, there is no single universally utilised exergy methodology. A number of different methodological approaches have evolved over the past 30 years based on the same theoretical foundations. The choice of the appropriate technique depends on the characteristics of the system under consideration and the information that the investigator aims to collect. Conventional exergetic approaches have been widely used in the past to comment on the resource utilisation and energy transformation of industrial processes e.g. (Kotas, 1985; Szargut et al., 1988), economic sectors e.g. (Dincer et al., 2004; Ediger and Camdali, 2007; Xi and Chen, 2006) and nations e.g. (Chen et al., 2006; Ertesvag, 2001).

In this paper three emerging exergy methodologies that in certain ways have gone beyond the conventional exergy analysis and in our view offer potential for sustainability planning and assessment are discussed in depth; Extended Exergy Analysis (EEA), Ecological Cumulative Exergy Consumption (ECEC) and eco-exergy.

EEA attempts to provide a picture of how human driven processes are interacting with the biosphere and the society. It was designed to make use of the correlations that exists between exergy and economic value by providing a theory of value similar to that of emergy synthesis (Milia and Sciubba, 2006; Sciubba 2001, 2003a,b). Material, energy, waste and atmospheric emission flows are calculated following the conventional exergy analysis. The novelty of the EEA over conventional exergy analysis, however, lies in the fact that it addresses and quantifies in thermodynamic terms capital flows and labour. The exergetic equivalent of a capital flow is calculated by multiplying the cash flow with an exergy-to-money ratio which is calculated in a similar manner as the emergy-to-money ratio (refer to the previous section) by dividing the overall exergy inputs into a country with an estimate of monetary circulation for that given country such as broad money. Labour flows are calculated following a similar procedure.

However, despite its novelties EEA like other exergy approaches still fails to take into consideration most of the "free" ecosystem services (apart from pollution abatement) such as rain and carbon sequestration that play an important role for human societies. ECEC analysis tends to bridge that gap by accounting for the free ecological processes through using certain emergy derived procedures. Even though the technique was initially limited to industrial processes it has lately been applied to other systems (Ukidwe and Bakshi, 2007). As EEA, ECEC has been influenced to a great extent from emergy analysis not the least by adopting concepts such as Transformity while "…ECEC becomes equivalent to emergy if the analysis boundary allocation approach and method for combining global energy inputs are identical" (Hau and Bakshi, 2004). The link between ECEC and emergy is more obvious if one looks the quantification procedure closer. For example, Ukidwe and Bakshi's (2007) input–output model that was based on ECEC principles identified and quantified:

- ecological inputs (products and services). The exergetic content of products such as metals, minerals and fuels is derived through a conventional exergy analysis and subsequently is multiplied by the relevant emergy transformities. Ecosystem services are quantified following standard emergy synthesis. It is worth mentioning here that only supply based ecosystem services which are measured solely in biophysical terms are quantified. Value-based ecological services such as those relevant to cultural and recreational uses which depend on people perceptions are not included;
- human inputs (labour). The ECEC equivalent of human labour is calculated through the Transformity of labour that can be found in the emergy literature;
- impact of emissions on human health. Pollutant emissions are quantified in terms of Disability Adjusted Life Years (DALYs) that considers several important categories such as respiratory disorders, climate change etc. Generally speaking there is a linear relationship between DALY and ECEC where 1 DALY equals a standard ECEC value that varies from country to country. A comprehensive explanation of the conversion between DALY and ECEC is included in (Ukidwe and Bakshi, 2004).

As it has been highlighted there is a strong connection between ECEC and emergy synthesis. Perhaps the most important methodological difference between the two is the different algebra they use especially in the case of allocation of multiple outputs. In particular allocation decisions in ECEC is a subjective decision as in the case of the Life Cycle Assessment (LCA) (Ukidwe and Bakshi, 2007). Other conceptual differences include the disregard of other controversial concepts such as the Maximum Empower Principle and the much moderate stance on biophysical value and its relation to economic value (refer to Sections 3.1 and 3.5).

Eco-exergy has been developed by S.E. Jorgensen over a period of 30 years (Jorgensen and Mejer, 1977) and is the most distinctive approach in the exergetic family of techniques. So far it has been mainly applied to the study of natural and especially aquatic ecosystems e.g. (Jorgensen, 2007; Salas et al., 2005) What makes eco-exergy different from other exergetic approaches, is the fact that it assesses and quantifies the exergetic content of the information coded by different entities, either animate or inanimate as well as the damage induced by anthropogenic exploitation of an ecosystem. The most obvious point of divergence from conventional exergy, EEA and ECEC discussed earlier is that it makes use of a different calculation procedure and of a different reference state. The departure from the reference states usually used by the techniques discussed earlier stems from the different focus of eco-exergy resulting in the conventional reference states not being practical for use in an ecosystem context (Jorgensen, 2007). From this starting point the eco-exergy of an ecosystem can be defined as the work which the ecosystem can perform relative to the same ecosystem at the same temperature and pressure but at thermodynamic equilibrium, where there is no gradient and all components are inorganic at the highest possible oxidation (ibid). Generally speaking, the higher the ratio of the embodied to the biomass exergy, the greater the distance from equilibrium or in other words, the distance from the death of the system, and thus the greater the ecosystem's health (Bendoricchio and Jorgensen, 1997; Verdesca et al., 2006). To put it simply the higher the amount and diversity of biomass in an ecosystem the higher the amount of information contained in it and thus the higher its eco-exergy. Eco-exergy has been proposed as a holistic indicator of ecosystem health and sustainability (Jorgensen, 2006a,b, 2007; Jorgensen and Nielsen, 2007). Finally, eco-exergy has recently been applied and in larger human systems such as countries (Jorgensen, 2006a).

3.3. Ecological Footprint

The Ecological Footprint (EF) was developed by W. E. Rees and M. Wackernagel at the beginning of the 1990s. Even though there have been several footprinting methodologies in the past (Deutsch et al., 2000), this particular approach has gained the most prominence.

There are several definitions of the concept in the literature but two definitions that best convey its meaning have been provided by its developers describing the EF both as a metric and as a tool;

"…the total area of productive land and water ecosystems required to produce the resources that the population consumes and assimilate the wastes that the population produces, wherever on Earth that land and water may be located." (Rees and Wackernagel, 1996)

"…an accounting tool that estimates the resource consumption and waste assimilation requirements of a defined human population or economy in terms of a corresponding productive land area." (Wackernagel and Rees, 1996)

The EF is regarded as a simple yet intuitive approach for investigating the demand of a given population's (individual, household, city, region, nation, planet) or a product's/service's on aspects of natural capital as well as its supply for that population. Its methodology relies on the formulation and comparison of two distinct accounts for any population, the biocapacity account and the footprint account or to put it in a different manner the ecological supply account and the human demand account. Both these are measured with a common unit of measurement, the global hectare (gha), making their comparison feasible.

So far two distinct methodologies have been used for its calculation; the component method (mainly at the sub-national level) and the compound-based method (mainly at the national and global level). In the component method the amount of all goods and services that a population consumes is identified and subsequently assessed through extensive Life Cycle Assessments (LCA). On the other hand the compound method makes use of better quality aggregated national data. In other words the component-based approach has the characteristics of a "bottom-up" approach while the compound-based those of a "top-down" (Chambers and Lewis, 2001; Simmons et al., 2000).

However, despite their differences the two methodologies share a common theoretical core as in the centre of both methodologies according to Wackernagel et al. (1999) lie the assumptions that it is possible to keep track of all the materials and human services required to sustain a human population and that most of these inputs can be converted to a corresponding biologically productive area.

In the footprint's calculation the different land types have been divided into six categories; built-up land, energy/CO_2 area, cropland, grazing land, forest and fishing ground based on the assumption that each land type provides for mutually exclusive demands on the biosphere while their sum equals the total EF. Subsequently the demand for each of these types of land for a number of different human activities such as food, shelter, mobility, goods and services is calculated.

Two key concepts incorporated into the EF calculations are those of the equivalence factor and the yield factor. Wackernagel et al. (2005), state that the former represents the potential productivity of a given bio-productive area relative to the world's average potential productivity for all bio-productive areas. The yield factors on the other hand describe the extent to which a biologically productive area within the system under study is more (or less) productive than the global average (ibid).

3.4. Attractive features and criticisms

The three biophysical approaches have been used in diverse case studies. This increasing popularity has led to a large number of peer reviewed articles exposing their assumptions and commenting on their perceived strengths and weaknesses. A collection of the most relevant attractive features and criticisms is included in Table 2. For more detailed information the reader is directed to more comprehensive publications (van den Bergh and Verbruggen, 1999; Brown and Herendeen, 1996; Cleveland, 2005; Cleveland et al., 2000; Ecological Economics, 2000; Herendeen, 2004; IVM, 2002; Mansson and McGlade, 1993; Sciubba and Ulgiati, 2005; Valero, 2006).

As already mentioned a sustainability metric, in order to be integrated it must be capable of addressing/ quantifying a wide variety of environmental, social and economic issues. There is no doubt that the list of such issues that influence urban sustainability is long. Some of the most important sustainability issues have been included in Tables 3–5 which also shows each biophysical metric's ability to quantify them. The issues were selected from various indicator lists (UN, 2001, UN-HABITAT, 2004; WHO, 2004) and can be viewed as a concise but not comprehensive, list of the most significant issues that planners and stakeholders usually consider. As can be seen in Tables 3–5 biophysical tools are quite successful in capturing certain environmental and economic issues. However, it should be noted that given the different concept of value that they employ (refer to the next section) they do not address economic issues from the same perspective as conventional economic analysis does. Furthermore, biophysical tools seem quite unable to consider most social issues.

Table 3
Economic issues addressed by the biophysical approaches

	EEA	Eco-Exergy	ECEC	Emergy	Ecological Footprint
Cost (Materials, energy, water, information, services, social etc)	√ [a, e]	√ [b, e]	√ [a, e]	√ [c, e]	√ [d, e]
Income	√ [e]	X	X	√ [e]	X
Labour	√	X	√	√	X
Efficiency (process)	√	X	√	√	√
Production/ Consumption Patterns	√	√	√	√	√
Asset Replacement	X	X	X	√	X
Trade	√	X	X	√	√
Debt (Internal and External)	X	X	X	√	X

a) Not for social and information cost.
b) Can capture material, energy and information costs.
c) Social cost has only rarely been evaluated e.g. (Kang and Park, 2002).
d) Social and Information cost cannot be quantified. Cost for certain other categories is indirectly calculated through the bio-capacity accounts; e.g. water.
e) Not in the traditional economic sense as they employ a different concept of value (refer to Section 3.5).

3.5. Some comments

All three biophysical approaches have a similar aim: to explain the relationships within complex systems through a natural science perspective. Nevertheless, apart from being completely different paradigms they share some common characteristics. In the bibliography one can also find hybrid methodologies that couple emergy, exergy and the ecological footprint (Chen and Chen, 2007; Hau and Bakshi, 2004; Zhao et al., 2005).

The most fundamental point of convergence is that the three biophysical models reviewed follow the spirit of strong sustainability without being direct measures of it (Neumayer, 2003). For example Wackernagel et al. (2005) state that the since the Ecological Footprint is rooted in the concept of ecological overshoot it "…tracks core requirements for strong sustainability and identifies priority areas for weak sustainability". Bastianoni et al. (2005) have also argued that certain indicators based on exergy can be linked to the concept of strong sustainability.

Furthermore, all three employ a common concept of value that is radically different from the one employed in economics. To put it simply, their underlying philosophies come in contrast to those of traditional economic analysis as they employ a "cost of production theory of value" (Patterson, 1998) and are essentially based on a donor system of valuation as Odum (1996) designated. For example, Brown and Ulgiati (1999) state that value in emergy synthesis is assigned based on how much energy, time, effort, materials etc has been invested in a product or service, something completely different from the traditional economics' donor-receiver system where value according to Pearce (1993b) is partly determined by public's WTP, and partly by the scarcity of the product/ service. Similarly in Extended Exergy Analysis value is assigned based on how much exergy (in the form of materials, labour, capital etc) has been embedded in a commodity (Sciubba, 2003a). Finally value in the Ecological Footprint is assigned according to how much bioproductive land must be appropriated by a given population in order to produce the product/services that

Table 4
Environmental issues addressed by the biophysical approaches

	EEA	Eco-Exergy	ECEC	Emergy	Ecological Footprint
Ecological health and quality (e.g. biodiversity, soil erosion, acidification, water quality, etc)	X	√ [f]	X	√ [g]	X [h]
Ecological Services	some aspects [i]	X	√	√	some aspects [j]
Depletion of natural capital	√	√	√	√	some aspects [k]
Pollution and waste	√	X	√	√	some aspects [l]
Water consumption	√	X	√	√	X [m]

f) Accounts only in a holistic manner and not for specific aspects of it such as biodiversity, etc e.g. (Jorgensen, 2006b).

g) Can account in a holistic manner as well as for specific aspects of it, e.g. biodiversity (Brown and Bardi, 2001), nutrients (ibid) etc.

h) Quantified indirectly. Some of the specific issues linked with ecosystem health (e.g. deforestation, acidification etc) lower the carrying capacity of the system under investigation so it renders the attainment of sustainability more challenging. It must be noted that in order for a system to be sustainable carrying capacity must be grater than the ecological footprint.

i) Only for pollution abatement.

j) Only carbon sequestration is accounted for directly. Other services such as soil degradation are considered indirectly because loss of such a service results to either a lower bio-capacity account or to a higher EF (e.g. increased insecticide and fertiliser use to compensate for the lost environmental service).

k) Only aspects that can be readily translated to bio-productive land. Forests are easily accounted for. Trends towards fossil fuel depletion are also visible since increase of energy land over time means accelerated fossil fuel depletion. Depletion of minerals and other non-renewable resources is only partially accounted for or can be conducted through exergy analysis (Hong and Yamamoto, 2007).

l) Usually only CO_2 is accounted for. A few studies have taken account of other greenhouse gases (Lenzen and Murray, 2001). Other important pollutants including air pollutants (PM, NO*x*, SO2 etc), Persistent Organic Pollutants (POPs), radionuclides, heavy metals etc cannot be accounted for in the present form of the EF.

m) Indirectly through the biocapacity accounts.

it consumes and to assimilate its wastes. Thus one can safely conclude that the three biophysical approaches, and in fact all other biophysical approaches (Winkler, 2006) share a common concept of value and address sustainability through an "ecocentric" prism as they employ the perspective of the donor (ecosystem) during the valuation of the different sustainability issues. In our view economic valuation tends to be anthropocentric in the sense that value is assigned based on the perspective of the receiver, human population, and the effect on its utility. In a way, ecocentric valuation techniques can be viewed as more "objective" when compared with monetary valuations that rely heavily on subjective WTP/WTA scenarios that can be misleading in some cases as discussed in Section 2. However, this deliberate ignorance of human preference has also been criticised, by Cleveland et al. (2000) among others. A more detailed discussion on these different concepts of value and their implications for sustainability assessments can be found elsewhere (Gasparatos et al., 2007).

A second common characteristic is that all three tools can be used in certain cases as precautionary principle tools. A clear illustration of the precautionary principle potential of the three biophysical approaches is the way that greenhouse gas emission is quantified

Table 5
Social issues addressed by the biophysical approaches

	EEA	Eco-Exergy	ECEC	Emergy	Ecological Footprint
Culture and heritage	X	X	X	√	X
Education and training	few studies [n]	X	X	√	X
Employment	X	X	X	X	X
Human Health (e.g. exposure to pollutants and noise, infant mortality, life expectancy etc)	X	X	X [o]	X	X
Crime	X	X	X	X	X
Equity (gender, income, intragenerational etc) [p]	X	X	X	X	X
Peace and order	X	X	X	X	few studies [q]
Justice	X	X	X	X	X

n) e.g. (Belli and Sciubba, 2001).

o) considers only effects of pollution on human health e.g. (Ukidwe and Bakshi, 2004, 2007).

p) some aspects , refer to Section 3.5.

q) e.g. (Brown, 1977).

especially considering the current scientific uncertainty on climate change (IPCC, 2001). Essentially all three techniques account for minimum environmental impact by "demanding" zero emission to the environment. For example, in emergy synthesis pollution is addressed through the emergy investment needed to prevent it or to take care of waste in a sustainable manner or to abate the potential environmental damage. Extended Exergy Analysis (EEA) accounts for the total invested exergy by society and ecosystems to render pollutants harmless to the environment. However, the concept of the precautionary principle is most clearly illustrated through the ecological footprint and its energy component where CO_2 emissions are accounted through the forest area needed to sequester them. This area is calculated in a way to account for all CO_2 emitted by a population.

Furthermore biophysical tools can capture certain aspects of equity rather qualitatively. For example, by accounting for the resources used by a given population as well as the rate of their use one can gain an idea on whether there will be sufficient resources left to satisfy the potential needs of future generations. Furthermore by finding a per capita expression of the resources used presently one can compare resource consumption ("real wealth") between members of the present societies. Another interesting example is the application of emergy synthesis to highlight unequal trade between nations eg. (Odum, 1996, Ulgiati et al., 1994). Trade between nations that have different emergy-to-money ratios would result in the country with a higher emergy-to-money ratio to giving more emergy than it actually receives as payment from the state with a lower emergy-to-money ratio. Similar trade applications could also be conducted and for EEA and ECEC that share certain similar assumptions with emergy synthesis.

Another similarity between emergy synthesis and the ecological footprint is that integral parts of their methodologies have been criticised in the past, more specifically the concepts of transformity and the equivalence/yield factors. Both these concepts introduce the concept of quality; energy quality in the former case and quality of the bioproductive land in the latter case. On the other hand exergy analysis has not faced similar criticisms probably due to its strong and universally accepted methodological foundations.

All three biophysical approaches are quite robust and have been used for the investigation and comparison of alternative designs in case studies as diverse as residential heating systems (Zmeureanu and Wu, 2007), waste management (Sciubba, 2003b; Yang et al., 2003), wetland management (Ton et al., 1998), cropping systems (Lefroy and Rydberg, 2003) and fuels (Holden and Hoyer, 2005) just to name a few. This flexibility and ability to investigate different development options can render them invaluable tools for sustainability assessment and planning.

4. Sustainability indicators and composite indices

4.1. Indicators

Indicators have been used in many fields of human knowledge and endeavour in order to measure, assess and plan different actions and phenomena. It is no wonder that they have also became central to the Sustainable Development debate with a large number of national and international organisations and academic institutions conducting significant research in the field. The reasons for their popularity are that indicators

> "…can provide crucial guidance for decision-making in a variety of ways. They can translate physical and social science knowledge into manageable units of information that can facilitate the decision-making process. They can help to measure and calibrate progress towards Sustainable Development goals. They can provide an early warning, sounding the alarm in time to prevent economic, social and environmental damage. They are also important tools to communicate ideas, thoughts and values because as one authority said, "We measure what we value, and value what we measure." (UN, 2001).

Gallopin (1997) has documented a number of different meanings that have been assigned to the concept including variable, parameter, measure, value and empirical model of reality amongst others. A definition of indicator based on Gallopin (1996) is the one that will be used for the rest of the paper. In this definition an indicator, like a variable, is an operational representation of an attribute of a system. The value of these variables are the actual measurements or observations at different times/locations/populations/etc or combinations of these. Immediately it becomes evident that there is not a single indicator (e.g. GDP) that manages to capture all aspects of complex concepts such as Sustainable Development or a complex system such as a nation or a city. However a collection of indicators chosen and analysed under certain criteria will better describe such complex concepts and systems.

4.2. Composite indices

The composite indicator/index (CI) can be simply defined as an aggregation of different indicators under a well developed and pre-determined methodology. Thus

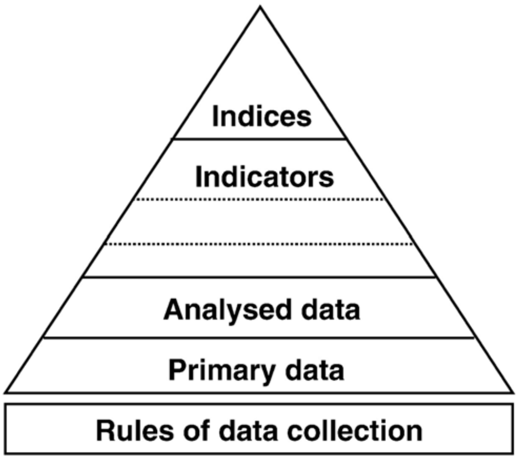

Fig. 1. The Information Pyramid (Hammond et al., 1995).

the CI lies on the top of an "Information Pyramid" (Hammond et al., 1995) as illustrated in Fig. 1. The aggregation of the diverse indicators in a single CI, differs conceptually from other indicator based techniques such as Multi-Criteria Assessments (MCA) where the constituent indicators are not aggregated into a single number. CIs are becoming increasingly popular for sustainability assessments at various scales e.g. (van Dijk and Mingshun, 2006; Krajnc and Glavic, 2005; Prescott-Allen, 2001).

CIs can be divided into several different categories according to the various methodological choices during their formulation. Perhaps the most relevant classification is the one between data driven and theory driven CIs (Niemeijer, 2002). Data driven (or bottom-up) approaches are preferred when data availability is the central issue concerning the development of the CI and high quality data is provided for all selected indicators. On the other hand theory-driven (or top-down) approaches are used when selecting the best possible indicators to accommodate a CI from a theoretical point of view while data availability is only one of the many aspects considered. One can argue that there can be a third category of CIs; the policy driven CIs or in other words the indices constructed especially for the monitoring of a certain policy. Whether data, theory or policy driven, CIs are powerful and communicative tools that can be of significant aid to planners and decision makers provided all methodological choices are transparent. The main attractive features and criticisms of the CIs are summarised in Table 6.

Accuracy and uncertainty are major issues that need to be considered both by the designer of the CI as well by the decision maker. In order to achieve the desired transparency and quality Nardo et al. (2005a) have designated a number of methodological steps for the construction of a CI. In this framework the main methodological steps that ought to be followed are the development of a theoretical framework, the selection of relevant indicators and data, a multivariate analysis, the imputation of missing data, a normalisation procedure, weighting and aggregation, an uncertainty and sensitivity analysis, the correlation of the CI with other published indicators and the visualisation of the findings for better communication of the results.

Nardo et al. (2005a,b) provide a comprehensive list of the different tools that can be used in each step of the procedure. Even though all steps are important for the quality of the final CI, a specific step, weighting and aggregation, seems to have been the focus of an ongoing debate and will be further discussed. A simple way to understand the significance of this step is by realising that at the end of this step a weighted single number is derived from different non-weighted indicator values.

A relevant quote on the issue of weighting is included in a popular scientific essay "…it is when hidden decisions are made explicit that the arguments begin. The problem for the years ahead is to work out an acceptable theory of weighting" (Hardin, 1968). Weights are essentially value judgements and thus "…greater weight should be given to components which are considered to be more significant in the context of the particular composite indicator" (OECD,

Table 6
Attractive features and criticisms of CIs (Nardo et al., 2005a)

Attractive features	Criticisms
Can summarise complex or multidimensional issues.	May send misleading policy messages if they are poorly constructed.
Easier to interpret.	May invite simplistic policy conclusions.
Facilitate the task of ranking alternatives over time on complex issues.	May be misused, e.g., to support a desired policy, if the construction is not transparent and lacks sound statistical or conceptual principles.
Reduce the size of a set of indicators or include more information within the existing size limit.	The selection of indicators and weights could be the target of political challenge.
Place issues of performance and progress at the centre of the policy arena.	May disguise serious failings in some dimensions and increase the difficulty of identifying proper remedial action.
Facilitate communication with the general public and promote accountability.	May lead to inappropriate policies if dimensions of performance that are difficult to measure are ignored.

2003). There are several different methodologies for the extraction of weights both participatory and non-participatory documented in the literature. Techniques such as the Analytical Hierarchy Process (AHP), budget allocation and Conjoint Analysis (CA) are participatory processes where selected participants (public and/or experts) are directly asked to state their opinion on the importance of the sustainability issues represented in the composite index. Other weighing techniques do not involve stakeholders and they rely more on statistics in order to produce the weights, e.g. Benefit of the Doubt (BOD). The interested reader is referred to Nardo et al. (2005a,b) for a description of these tools.

However weights do not always retain this status within a CI. Munda and Nardo, (2005a) have shown that by using linear aggregation as an aggregation technique the assigned weights end up gaining a trade-off status that implies complete compensability and substitutability between the components, i.e. indicators, of the CI. In other words, in a CI constructed using a simple weighting and linear aggregation procedure higher performance of an indicator X (e.g. economic activity) has the ability to compensate for lower performance in an indicator Y (e.g. depletion of natural resources). This bears a striking resemblance to some of the criticisms of CBA discussed in Section 2.4. A classic example and discussion on the topic, albeit more intuitive than the mathematical approach by Munda and Nardo (2005a), is the case of the Human Development Index (HDI) as presented by Sagar and Najam (1998) and Desai (1991) amongst others. A way to overcome this situation is to adopt either different aggregation techniques such as geometric aggregation or even not to aggregate at all. But even by choosing a geometric aggregation there is still a degree of compensability between the different indicators although it is smaller than the linearly aggregated CI (Nardo et al., 2005a). A recent comparative study between three commonly used aggregation techniques (linear, geometric, weighed displaced ideal) concluded that in most cases the geometric aggregation technique results in the minimum loss of information (Zhou et al., 2006).

4.3. Some comments

A significant advantage of indicators over all other approaches reviewed in this paper is their increased accuracy in evaluating/quantifying the different sustainability issues under consideration since these issues need not be translated to other metrics such as money. However the composite index is influenced to a great extent by the choice of the indicators. For example Lee (2006) points out that if the aim of the analyst is to address inter-generational equity then stock indicators would be more appropriate. Conversely in situations that the analyst seeks to capture intra-generational equity then a choice of distributional indicators would seem more justified.

Furthermore, the compensability between the components of the CI implies the existence of trade-offs and renders aggregated CI weak sustainability tools in the sense discussed in Section 2. This is not necessarily a disadvantage but it is a feature of the methodology that planners and stakeholders need to be aware of during the planning and decision making process. Considerations of the acceptability of trade offs must influence the choice of a proper aggregation methodology with linear aggregation being chosen in cases where unlimited substitution is desirable, geometric aggregation in cases of limited substitution and MCA being chosen when trade-offs are considered to be inappropriate. Arrow's impossibility theorem ruling out the existence of a perfect aggregation technique for ranking alternative options (e.g. alternative designs, policies etc) is also worth noting here (Arrow, 1963) as quoted by Munda and Nardo, 2005b.

Methodological choices in our opinion also affect the extent to which a composite index considers the precautionary principle. For that choice of the appropriate indicators seems to be quite important. Choosing indicators that represent issues which may have a detrimental effect on human and ecosystem health ensures that they are being taken into account during the decision making process. Another way of entering precautionary principle considerations to the composite index could be to assign greater weights to indicators that represent such issues (refer to Section 2.4 for a similar discussion on the CBA). Finally the normalisation of the indicators can also play a major role.

5. The tools within a sustainability assessment framework

Pope et al. (2004) distinguish between two existing approaches to Sustainability Assessment; Environmental Impact Assessment-led (EIA-led) or Baseline-led and Objectives-led. The former is characterized by a conscious attempt to minimize adverse sustainability impacts by preventing a project/policy to be undertaken if it is expected to lead towards greater future unsustainability. In particular "…this approach to sustainability assessment aims to ensure that impacts are not unacceptably negative overall, meaning that the guiding acceptability criterion for a proposal is that it does not lead to a less sustainable outcome" (ibid). In order for Baseline-led assessments to qualify as sustainability assessments it must be ensured by the assessors that the test of approval is the net improvement beyond the current baselines.

Gibson (2000) indicates that an inherent implication of this approach is the fact that it allows for trade-offs between different sustainability issues following a weak sustainability perspective.

In Objectives-led integrated sustainability assessment on the other hand the central aim is to achieve a certain sustainability vision and to capture "…the extent to which the implementation of a proposal contributes to this vision" (Pope et al., 2004). This comes in contrast to the Baseline-led approach which focuses on whether the environmental/social/economic etc impacts are acceptable compared to a certain baseline. Monetary tools, biophysical models and composite sustainability indices have characteristics that seem compatible with both types of sustainability assessments and in some cases it is not clear which tool is most appropriate for each type of assessment.

For example it has been mentioned that CBA is a weak sustainability tool in the sense that it identifies whether a project/policy should be undertaken by comparing relevant costs/benefits and allowing trade-offs between sustainability issues. Thus it can be argued that CBA is quite a sound option for Baseline-led sustainability assessments. On the other hand CBAs attempt to identify the project that results in the greatest welfare improvement for the affected stakeholders (both present and future). This conscious attempt to maximize a specific sustainability objective, economic welfare in this case, seems to be aligned with the rationale of Objective-led sustainability assessments. Nevertheless CBA does not employ specific sustainability targets; e.g. CBA analysts have refrained in the past, quite justifiably in our opinion, from designating an optimum cost/benefit ratio to be considered as a legitimate sustainability objective. As a result CBA seems to be a sounder methodological option for Baseline-led sustainability assessments. Similar conclusions can be drawn for the ISEW.

As has already been mentioned biophysical models attempt to capture the environmental support required by a project/policy. Central to their philosophy, albeit not overtly evident, is their vision of minimizing human impact on the biosphere. For example when the effect on overall sustainability of two projects/policies is assessed through a biophysical model then the project/policy with the lowest EF or non-renewable emergy/exergy consumption would seem to be a more acceptable option as this would imply lower resource utilization and in some cases a lower environmental impact. This inherent vision of minimizing human impact implies a close connection with both Baseline-led and Objective-led sustainability assessments. In our opinion the biophysical models have greater potential as part of an Objectives-led sustainability assessment. However, an important issue arising is that biophysical models have not articulated, with a few exceptions, clearly defined sustainability objectives. For example the project/policy with the lowest ecological footprint would be considered as the most sustainable but in most cases a specific target has not been articulated. Emergy synthesis seems to have been an exception amongst the biophysical models with a string of indicators having been developed to highlight sustainability objectives. Sustainability objectives such as the Emergy Sustainability Index for example have been under scrutiny as it has been shown to neglect the role of local non-renewable resources (Bastianoni et al., 2007). Furthermore given that biophysical models are not able to capture certain economic and social issues, their findings should be supplemented with findings from other tools. It must be stressed here that significant importance should also be paid to the interrelations between different sustainability issues (George, 2001).

Composite indices are quite flexible sustainability assessment tools and according to the methodological choices during their construction they can be accommodated in either type of sustainability assessment. Choice of an aggregation technique greatly influences whether the composite index acts as a strong or weak sustainability tool. Furthermore, choice of a normalization technique can have similar results. For example one of the most usual normalization techniques is distance to a reference (Nardo et al., 2005a). If that reference is a collection of legislative limits then the normalized values within the composite sustainability index will provide a baseline below which the sustainability of the project/policy can be deemed unacceptable. On the other hand if the reference is chosen as the best present practice or a state which can actually be considered sustainable then the composite index would act as a sustainability objective that the project/policy should aspire to achieve. Thus in the former case a sustainability index could be used within a Baseline-led sustainability assessment while in the latter case within an Objectives-led sustainability assessment.

It should be noted here that in addition to the Baseline and Objective-led sustainability assessments there is also a "hybrid type of sustainability assessment" that aims to identify the best project/policy from a comprehensive sustainability perspective by maximizing the multiple reinforcing net benefits through selection, design and adaptive implementation of the most desirable option while avoiding significant losses (Gibson et al., 2005; Gibson, 2006). In the author's view none of the tools described earlier should be the sole sustainability evaluation tool for hybrid sustainability assessments for the following reasons. Firstly, both biophysical and monetary tools adopt different perspectives during the evaluation of the various sustainability impacts as discussed in Section 3.5. Moreover, biophysical and monetary tools employ different sustainability visions. The adoption of any single family of these tools means that by default a certain perspective will be overrepresented in the sustainability assessment, ruling out a comprehensive sustainability viewpoint. However, adoption of a combination of biophysical and monetary tools may result in a more comprehensive sustainability perspective during the assessment which is in the spirit of the "hybrid type of sustainability assessment". Integrating the outputs of such a pluralistic approach is complicated and further research is required. A second characteristic of this "hybrid type of sustainability assessment" is the conscious attempt to discourage trade-offs to the greatest extent possible (Gibson et al., 2005; Gibson, 2006). However weak sustainability tools such as CBA, ISEW and composite indices (depending on the aggregation methodology employed) allow for trade-offs between the different sustainability issues as discussed in Sections 2.4 and 4.2.

6. General discussion

Table 7 contains a summary of the extent to which the reviewed tools embrace our five criteria for holism. Most of the tools reviewed in this paper are quite flexible and able to consider several economic, environmental and social sustainability issues. Apart from the ISEW that contains a standard set of sustainability issues, all other tools depend on the ability of the analysts/stakeholders to choose the issues that seem to be appropriate in the context of their study. Furthermore all tools, with the exception of the ISEW, are able to consider and predict the future impacts under different development scenarios. The spatial and temporal boundaries can be designated by the analysts/stakeholders but extended boundaries are expected to result in high uncertainties which can have significant implications for the decision making process. When it comes to the precautionary principle, monetary tools are unable to address it. They seem to assume certainty by using monetised values (the exact effect of a sustainability impact on human utility might be very uncertain or even unknown) where, by definition, certainty does not exist. Furthermore they usually utilize the same weights for cost/benefits which imply a risk neutrality perspective rather than risk aversion. Composite indices can consider aspects of the precautionary principle subject to certain methodological choices especially during the indicator choice, weighting and normalisation procedures. Biophysical tools can also capture certain aspects of the precautionary principle as illustrated with the case of climate change discussed in Section 3.5. Some monetary valuation tools are participatory in the sense that they capture the public's stated preference over certain sustainability issues in monetary terms. For example in CVM the public is directly asked about its WTP/WTA over different sustainability issues. Composite indices can also be participatory tools depending on methodological choices. Steps in which the public can be involved include the choice and the weighing of the indicators that will constitute the composite index. Biophysical tools and the ISEW lack any explicit steps in which the public can be involved. Whether CBA can address equity is debatable given that CBA is based on economic efficiency rather than equity considerations. Choices within CBA that seem incompatible with inter and intra generational equity are the Kaldor–Hicks criterion for aggregation and the discounting of future costs/benefits respectively. The ISEW can to a certain extent capture intergenerational equity since it starts from an index of income inequality. Intra-generational equity is considered somewhat intuitively through a time series of the ISEW. Continuous decrease of the ISEW over time implies a worse quality of life over time. Biophysical tools can also give some indirect information on equity issues as discussed in Section 3.5.

Comment could be made and on the diffusion of the different techniques between practitioners. Monetary and indicator tools seem to have been more widely diffused among practitioners probably as a result of their simplicity, widespread acceptance and easy communication to the stakeholders. Of the biophysical approaches only the ecological footprint seems to be entering the mainstream at

Table 7
Extent to which the reviewed tools embrace our criteria for holism

Criteria	CBA	ISEW	Biophysical	Composite indices
Integrated [a]	√	√	X (Tables 3–5)	√
Predictive [b]	√	X (Section 2.4)	√	√
Precautionary	X	X	Some aspects	Depends on method. choices
Participatory [c]	Depends on monetisation methodology	X	X	Depends on method. choices
Equity consideration	Debatable (Section 2.4)	Some aspects	Some aspects	Depends on method. choices

[a] Consider environmental, social and economic issues.
[b] Consider different development scenarios and compare their different outcomes in an informative manner.
[c] Engage with stakeholders in the assessment procedure.

the moment and probably due to similar reasons as "…land is a more familiar, acceptable, and motivating concept to most people than energy, CO_2 or biodiversity" (Herendeen, 2000) and to the amount of media attention it attracts. Of particular interest is the case of the UK where the development of a tool (Barrett and Simmons, 2003) for measuring the sustainability of local authorities by the two leading UK footprint consultancies is expected to give further boost to the practice. On the other hand, despite their strengths, the two thermodynamic approaches, emergy and exergy have not enjoyed similar acceptance outside small circles of supporters. Analysts and stakeholders need to become familiar with the methodologies and their associated assumptions in order for these tools to fulfil their potential.

However despite their advantages the authors perceive a growing disenchantment over the applicability of reductionist metrics and tools for sustainability assessment (Gasparatos et al., 2007). One such criticism comes from complexity theory and its view that systems such economies, ecosystems, societies and cities are complex and adaptive while their "…properties are not fully explained by an understanding of their constituent parts" (Gallagher and Appenzeller, 1999). Additionally, the emergence of post-normal science (Funtowicz and Ravetz, 1993) and the belief that it is essential to describe complex systems through the synthesis of their different, non-reducible and perfectly legitimate perspectives (Funtowicz and Ravetz, 1994) further adds to the discontent over the validity of reductionism in sustainability assessment. On this ground reductionism has been criticized in the past as inadequate for sustainability policy making e.g. (Munda, 2006) amongst others. Some of the proposals for assessing the progress towards sustainability in a non-reductionist manner are the different participatory sustainability assessment frameworks found in the literature e.g. (Banville et al., 1998; De Marchi et al., 2000; Gamboa, 2006; Gamboa and Munda, 2007; Guimarães Pereira et al., 2003; Kallis et al., 2006; Madlener and Stagl, 2005; Munda, 2004; Mustajoki et al., 2004; Salgado et al., 2006). These frameworks advocate stakeholder involvement in varying degrees and at various stages of the sustainability assessment and they make extensive use of multi-criteria methodologies for the designation of the most appropriate solution.

7. Conclusions

None of the metrics and tools discussed in this paper seems to be capable of assessing the progress towards sustainability in a holistic manner. The need to address the multitude of environmental, social and economic issues, together with intergenerational and intragenerational equity concerns, formulates problems that none of the above reductionist approaches can tackle individually in an adequate manner. Moreover the need to act with a precautionary bias and engage the public further perplexes a sustainability assessment. Another implication emerges from the fact that monetary and biophysical approaches view sustainability problems from different but complementary perspectives. Even though they provide complementary snapshots of the same picture it can be argued that they are unable to capture the whole picture. On the other hand, significant information is lost during the aggregation step in composite indices. As a result of the above the choice of metrics and tools must be informed according to the context and the characteristics that the analyst seeks to highlight. Both analysts and stakeholders need to be informed about the underlying assumptions of the tools used. It is worth repeating here that these methodological assumptions are not necessarily disadvantages but features that must be known in order to assure the transparency of the final outcome. Finally, because no single tool can encompass the complete range of perspectives that may be embraced by stakeholders there is merit in considering the use of a variety of tools the choice of which would be heavily conditioned by the context of the assessment. Integrating the outputs from such a pluralistic approach should be the subject of future research.

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