SPECULATIONS ON WEAK AND STRONG SUSTAINABILITY

by

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ABSTRACT

In this paper a number of salient elements of the Sustainable Development debate are identified - equity, inheritance asset portfolio, systems thinking and environmental ethics - in order to draw out the range of contrasting views that currently exist between economists, scientists and philosophers. Four basic sustainability 'paradigms' (probably overlapping) are identified and analysed. The case for a strong (but not very strong) sustainability position is advocated.

1. INTRODUCTION

According to one recent critique of "sustainability" thinking there is a proliferation of definitions of sustainability, but it remains far from clear what concept of "sustainable development" (SD) can be both morally acceptable and operationally meaningful (Beckerman, 1992). The most quoted principle of SD is that offered by the Brundtland Commission: "...development that meets the needs of the present without compromising the ability of future generations to meet their own needs (WCED, 1987). SD is a multi-faceted concept and one that has generated an extensive, on-going, multi-disciplinary debate.

From the conventional economic perspective, the sustainability issue has at its core the phenomenon of market failure and its correction via 'proper' resource pricing. What is required is an intertemporally efficient allocation of environmental resources through price corrections based on individual preference value e.g. Solow (1974), Solow (1986). A vast literature has therefore grown upon the various monetary valuation methods and techniques available to 'price' the range of environmental goods and services provided by the biosphere (i.e. market-adjusted, surrogate-market and simulated-market methods).

Critics of this standard economic position, however, have variously argued that:

1) economic sustainability is more a matter of intergenerational equity (e.g. Howarth and Norgaard, 1990; Howarth, 1991; Barbier, Markandya and Pearce, 1990; Pearce and Turner, 1990; and Turner, 1988b); and that incorporating environmental values per se into the policy-making process will not bring about sustainability unless each generation is committed to transferring to the next sufficient natural resources and capital assets to make development sustainable

(Howarth and Norgaard, 1992);

2) Conventional economic and ecological approaches to sustainability are largely disjoint, they address different phenomena. Intertemporal price efficiency is not a necessary condition for ecological sustainability (defined in terms of ecosystem stability and resilience and requiring the constraint that the allocation of economic resources should not result, via overall system feedbacks, in the instability of the economy-environment systems as a whole) and further that the pursuit of intertemporal efficiency on the basis of the individual preferences of the current generation may well be inconsistent with ecological sustainability (Common and Perrings, 1992);

3) An ethical shift away from, or modification of, traditional individualistic moral reasoning is required unless depletion of the moral capital stock is not to take place (Daly and Cobb, 1989; Common and Perrings, 1992; Norton, 1989);

4) Cultural diversity and biological diversity are both prerequisites for sustainable development and therefore the conservation of cultural capital and the fostering of local sustainable livelihoods are important from a policy point of view (e.g. Berkes and Folke, 1992; Chambers and Conway, 1992; Perrings, Folke and Mäler, 1992).

A number of salient elements of the SD debate can therefore be identified, in order to draw out the range of contrasting views that currently exist between economists, scientists and philosophers - see figure 1. We examine five such foci in the SD debate in succeeding sections.

2. INTRAGENERATIONAL AND INTERGENERATIONAL EQUITY

SD is future oriented in that it seeks to ensure that future generations are at least as well off, on a welfare basis, as current generations, it is therefore in economic terms a matter of intergenerational equity and not just efficiency. The distribution of rights and assets across generations determines whether the efficient allocation of resources sustains welfare across human generations (Howarth and Norgaard, 1992). The ethical argument is that future generations have the right to expect an inheritance sufficient to allow them the capacity to generate for themselves a level of welfare no less than that enjoyed by the current generation. What is required then is some sort of intergenerational social contract.

SD also has a poverty focus, which in one sense, is an extension of the intergenerational concern. Daly and Cobb (1989) have argued that families endure over intergenerational time. To the extent that any given individual is concerned about the welfare of his/her descendants, he/she should also be concerned about the welfare of all those in the present generation from whom the descendant will inherit. Accordingly, a concern for future generations should reinforce and not weaken the concern for current fairness. Ethical consistency demands (despite the trade-offs involved) that if future generations are to be left the means to secure equal or rising per capita welfare, the means to maintain and improve the wellbeing of today's poor must also be provided. Collective rather than individual action is required in order to effect these socially desirable intra- and intergenerational transfers. In any case, it seems to us, that no nation that neglects the most vulnerable in society ought to be labelled 'developed' or 'developing'.

The degree of concern, as expressed by the rate of time discount (DR) attached to the welfare of future generations, that is ethically required of the current generation is another controversial matter. Six positions seem possible - moral obligations to the future exist, but the welfare of the future is less important than present welfare ($0 < DR < \infty$); moral obligations to the future exist and the future's welfare is almost as important as present welfare (social time preference rate = STPDR; ($0 < STPDR < DR$); discounting procedure is only acceptable after the imposition of pre-emptive constraints on some forms of economic development; obligations to the future exist and the future is assigned more weight than the present (DR is negative); rights and interests of future people are exactly the same as contemporary people ($DR = 0$); there is no obligation all on the present to care about the future ($DR = \infty$).

3. SPECIFICATION OF THE SUSTAINABILITY INHERITANCE ASSET PORTFOLIO

Victor (1991) has recently remarked that one of the contributions that economists have made to the SD debate has been the idea that the depletion of environmental resources (source and sink resources) in pursuit of economic growth is akin to living off capital rather than income. SD is then defined as the maximum development that can be achieved without running down the capital assets of the nation, which are its resource base. The base is interpreted widely to encompass, man-made capital K_m, natural capital K_n, human capital K_h and moral (ethical) and cultural capital K_e and K_c. Victor identifies four 'schools' of thought on the 'environmental as capital' issue - the mainstream neoclassical school, the London school (after Pearce, Barbier, Markandya and Turner), the post-Keynsian school and the thermodynamic school (after Boulding, Georgescu-Roegen, Daly, Perrings and Common). Roughly speaking, this spectrum of views moves from a position we can label **'very weak sustainability'** through to one we call **'very strong sustainability'** (see also Klassen and Opschoor, 1990). Figure 2 formalises in simplistic fashion these various sustainability paradigms, which in practice are less clearly defined and are overlapping.

Figure 2: Sustainability Rules and Indicators

	NO CRITICAL NATURAL CAPITAL	CRITICAL NATURAL CAPITAL
VERY WEAK SUSTAINABILITY	$s/y - \partial K/y > o$	Perfect Substitution All K_n and K_m Growth Economy
WEAK SUSTAINABILITY	$s/y - \partial m/y - \partial n/y = WSI$	$WSI > 0$

	WSI > 0 $\lambda >$ _ n > Z	$\lambda >$ _ n > Z $\partial n^* \leq 0$
STRONG SUSTAINABILITY	$\partial n \leq 0$ WSI > 0	WSI > 0 $\partial n \leq 0$ $\partial n^* \leq 0$ $\partial K_c \leq 0$
VERY STRONG SUSTAINABILITY	Perfect Complementarity All K_n and K_m Stationary State Economy	WSI > 0 $\partial n \leq 0$ $\partial n^* \leq 0$ _ ≤ 0 $\partial K_c \leq 0$ $\partial K_e \leq 0$

Notes :

K = total capital assets
s = savings
∂m = depreciation on man-made capital
∂n = depreciation on natural capital
λ = technical change
_ = rate of population growth
n* = critical natural capital (no substitutes)
K_c = cultural capital
K_e = moral/ethical capital
Z = Lower bound stock limit (determined via SMS) to ensure ecosystem stability

3.1 Very Weak Sustainability (Solow-Sustainability)

This sustainability rule merely requires that the overall stock of capital assets ($K_m + K_n + K_h + K_e + K_c$) should remain constant over time. The rule is, however, consistent with any one asset being reduced as long as another capital asset is increased to compensate. This approach to sustainability is based on a Hicksian definition of income, the principle of constant consumption (buttressed by a Rawlsian maximin justice rule operating intergenerationally), production functions with complete substitution properties and the Hartwick rule governing the reinvestment of resource rents (Common and Perrings, 1992).

Thus, following Hicks, income is the maximum real consumption expenditure that leaves society as well endowed at the end of a period as at the start. The definition therefore presupposes the deduction of expenditures to compensate for the depreciation or degradation of the total capital asset base that is the source of the income generations, i.e. conservation of the value of the asset base. Assuming a homogeneous capital stock (perfect substitution possibilities) the Hartwick rule states that consumption may be held constant in the face of exhaustible resources only if rents deriving from the intertemporally efficient use of those resources are reinvested in reproducible capital.

It is now possible to derive an intuitive weak sustainability measure or indicator (in value terms) for determining whether a country is on or off a sustainable development path (Pearce and Atkinson, 1992). Thus a nation cannot be said to be sustainable if it fails to save enough to offset the depreciation of its capital assets. That is,

$$WSI > 0 \quad \text{if } S > \delta K \tag{1}$$

where WSI is a sustainability index, S is savings and δK is depreciation on capital. Dividing through by income (Y) we have,

$$WSI > 0 \text{ if } (S/Y) > (\delta K/Y) \tag{2}$$

or $$WSI > 0 \text{ if } (S/Y) > [\delta_m/Y + (\delta_n/Y)] \tag{3}$$

where δ_m is depreciation on man-made capital, δ_n is depreciation on natural capital and K_n and K_m are substitutable.

3.2 Weak Sustainability (Modified Solow-Sustainability)

Perrings (1991) and Common and Perrings (1992) highlight the fact that the technological assumptions (substitution possibilities) of the weak sustainability approach violate scientific understanding of the evolution of thermodynamic systems, and ecological thinking about the complementarity of resources in system structure and the importance of diversity in system resilience.

The London school has also modified the very weak sustainability approach by introducing an upper bound on the assimilate capacity assumption, as well as a lower bound on the level of K_n stocks that can support SD assumption, into the analysis (Barbier and Markandya, 1989; Pearce and Turner, 1990; Klassen and Opschoor, 1990). The concept of critical natural capital (e.g. keystone species and keystone processes) has also been introduced to account for the non-substitutability of certain types of (K_n) (e.g. environmental support services) and man-made capital (K_m). Thus the requirement for the conservation of the value of the capital stock has been buttressed by constraints aimed at the preservation of some proportion and/or components of K_n stock in physical terms.

The implications of this modified Solow-sustainability thinking seems to be the formulation of a sustainability constraint which will impose some degree of restriction on resource-using economic activities. The constraint will be required to maintain populations/ resource stocks within bounds thought to be consistent with ecosystem stability and resilience. To maintain the instrumental value (benefits) humans obtain from healthy ecosystems, the concern is not preservation of specific attributes of the ecological community but rather the management of the system to meet human needs, support species and geneity diversity, and enable the system to adapt (resilience) to changing conditions.

A set of physical indicators will be required in order to monitor and measure biodiversity and ecosystem resilience. As yet there is no scientific consensus over how biodiversity should be measured. Measuring genetic diversity presents the least difficulty but measuring species diversity is more problematic - measures of species richness, taxonomic richness, richness of genera or families have all been investigated and none are without difficulties. Problems associated with measuring biodiversity at the community level are even greater but such measures would be very useful to policy-makers if the aim is conservation at the ecosystem level. Community classification schemes can be developed from the global level, through biogeographic provinces, down to regions within a country. Ecoregion classification is based primarily on the physical environment. A minimum set of 22 indicators (including species richness, species risk index, community diversity etc.) has been proposed for wild and domesticated species (Reid et al., 1992).

For some commentators sustainability constraints of this type should be seen as expressions of the precautionary principle (O'Riordan, 1992) and one akin to safe minimum standards (SMS) (Bishop, 1978). Toman (1992) quoting the work of Norton has recently suggested that the SMS concept is a way of giving shape to the intergenerational social contract - see figure 3. Given irreversibility and uncertainty about the impact of economic activities on ecosystem performance SMS posits a socially determined dividing line between moral sustainability imperatives and the freeplay of resource tradeoffs (e.g. SS in figure 3). To satisfy the intergenerational social contract, the current generation might rule out in advance (depending on the social opportunity costs involved) actions that might result in damage impacts beyond a certain threshold of cost and irreversibility. Social and not individual preference values will be part of the SMS setting process (Turner, 1988b). Supporters of the technocentric paradigm might favour a line such as S_T-S_T while ecocentrics such as the Deep Ecologists might favour a line such as S_{DE}-S_{DE} (Pearce and Turner, 1990).

3.3 Strong Sustainability (Ecological Economics Approach)

As we have seen, the weaker versions of sustainability are consistent with a declining level of environmental quality and natural resource availability as long as other forms of capital are substituted for K_n, or the imposition of SMS is judged to impose to high a social opportunity cost given the inevitably uncertainties involved in conservation benefits forecasting.

A number of analysts, from a variety of disciplines, have drawn attention to the 'missing elements' in the economic calculus that underlies the weak sustainability rules. Many ecosystem functions and services can be adequately valued in economic terms, others may not be amenable to meaningful monetary valuation. Critics of conventional economics have argued that the full contribution of component species and processes to the aggregate life-support service provided by ecosystems has not been captured in economic values (Ehrlich and Ehrlich, 1992). Nor has the prior value of the aggregate ecosystem structure (life-support capacity) been taken into account in economic calculations, indeed it is probably not fully measurable in value terms at all. We examine this concept which we call **ecosystem primary value** in more detail in a later section. There is the risk therefore that as environmental degradation occurs, some life-support processes and functions will be systematically eroded, increasing the vulnerability (reduced stability and resilience) of the ecosystem to further shocks and stress.

On this strong sustainability view, it is not sufficient to just protect the overall level of capital, rather K_n must also be protected, because at least some of K_n (critical K_n) is non-substitutable. Thus the strong sustainability rule requires that K_n be constant, and the rule would be monitored and measured via physical indicators. The case for this 'strong' view is based on the combination of a number of factors - presence of uncertainty about ecosystem functioning and their total service value; presence of irreversibility in the context of some environmental resource degradation and/or loss; the loss aversion felt by many individuals when environmental degradation processes are at work; the criticality (non-substitutability) of some components of K_n; and finally, what Daly (1991) has termed the "scale effect", i.e. the scale of human impact relative to global carrying capacity. For him, the greenhouse effect, ozone layer depletion and acid rain all constitute evidence that we have already gone beyond a prudent "plimsoll line" for the scale of the macroeconomy.

While akin to SMS the strong sustainability (SS) rule is not the same since what is stressed in the latter approach is the combination of factors, irreversibility, uncertainty etc., not their presence in isolation. Further, SMS says conserve unless the benefits foregone (social opportunity costs) are very large. SS says, whatever the benefits foregone, K_n losses are unacceptable (i.e. constant "aggregate" K_n, not constant K_n for each asset).

SS need not imply a steady-state, stationary economy, but rather changing economic resource allocations over time which are not sufficient to affect the overall ecosystem parameters significantly, i.e. beyond the point where the stability (resilience) of the system or key components of that system are threatened. A certain degree of 'decoupling' of the economy from the environment should therefore be possible via technical change and environmental restoration investment in a 'moderated' growth scenario.

3.4 Very Strong Sustainability (Stationary State Sustainability)

The very strong sustainability perspective reduces to a call for a steady-state economic system based on thermodynamic limits and the constraints they impose on the overall scale of the macroeconomy. The rate of matter and energy throughput in the economy should be minimised. The second law of thermodynamics implies that 100% recycling is impossible (even if it were socially desirable) and the limited influx of solar energy poses an additional constraint on the sustainable level of production in an economy (the solar influx potential is a matter of some dispute). Zero economic growth and zero population growth are required for a zero increase in the "scale" of the macroeconomy. Supporters of the steady-state paradigm would, however, emphasise that 'development' is not precluded and that social preferences, community-regarding values and generalised obligations to future generations can all find full expression in the steady-state economy as it evolves (ie. conservation of the moral capital (K_e) on which economic activity eventually depends (Hirsch, 1976; Daly and Cobb, 1989).

4. A SYSTEMS AND CO-EVOLUTIONARY PERSPECTIVE FOR SUSTAINABILITY

The adoption of a systems perspective serves to re-emphasise the obvious but fundamental point that economic systems are underpinned by ecological systems and not vice versa. There is a dynamic interdependency between economy and ecosystem. The properties of biophysical systems are part of the constraints set which bound economic activity. The constraints set has its own internal dynamics which react to economic activity exploiting environmental assets (extraction, harvesting, waste disposal, non-consumptive users). Feedbacks then occur which influence economic and social relationships. The evolution of the economy and the evolution of the constraints set are interdependent, 'co-evolution' is thus a crucial concept (Norgaard, 1984; Common and Perrings, 1992).

Norton and Ulanowicz (1992) advocate a hierarchical approach to natural systems (which assumes that smaller subsystems change according to a faster dynamic than do larger encompassing systems) as a way of conceptualising problems of scale in determining biodiversity policy. For them, the goal of sustaining biological diversity over multiple human generations can only be achieved, if biodiversity policy is operated at the landscape level. The value of individual species, then, is mainly in their contribution to a larger dynamic and significant financial expenditure may not always be justified to save ecologically marginal species. Goal for policy should be to protect as many species as possible, but not all.

Ecosystem health (stability and resilience or creativity), interpreted in terms of an intuitive guide, is useful in that it helps focus attention on the larger systems in nature and away from the special interests of individuals and groups (Norton and Ulanowicz, 1992). The full range of public and private instrumental and non-instrumental values all depend on protection of the processes that support the health of larger-scale ecological systems. Thus when a wetland, for example, is disturbed or degraded, we need to look at the impacts of the disturbance on the larger level of the landscape. A successful policy will encourage a patchy landscape.

4.1 Environmental Resource Valuation

In particular from the weak sustainability perspective there is an essential link between sustainable development and monetary valuations of the environment in terms of willingness-to-pay (WTP) - see figure 1. The lack of meaningful monetary valuations of environmental assets would greatly circumscribe the weak sustainability case, based as it is on substitution possibilities between K_n and K_m.

Valuation techniques, notably the contingent valuation (CVM) approach, have been more extensively used, and many would argue have been substantially improved, during the 1980s. Recent debates, both within the economics profession and between economists and non-economists, about the 'usefulness' (reliability and validity) of CVM have brought to the surface several general and fundamental questions. The theory, methodology and application of non-market value measurement of environmental resources have all come under scrutiny. Harris and Brown (1992), for example, have noted that, who gains or loses from some environmental change and how feelings of responsibility and self-interest influence the value judgements of those gainers and losers, have important implications for CVM.

4.2 Debate Within Economics

The travel cost (TC) recreation demand model is the longest established indirect revealed preference approach, and of a great deal of empirical data has been accumulated using this approach. US practioners seems to be generally agreed that the TC methodology is successful and has produced meaningful value estimates.

Smith (1992) concludes that the estimates made using TC uphold rudimentary predictions of consumer theory (i.e quantity negatively related to their own price). Further, when TC is applied to comparable sites value estimates are broadly consistent, and when TC is applied to different types of recreation sites plausible differences in value have been revealed. The estimation of the value individuals place on changes in the recreation sites' quality features has, however, proved to be more problematic.

A somewhat different picture emerges if UK experiences are reviewed. After a short lived renewed interest in the TC method in the mid 1980s, there has since been a pronounced loss of confidence in TC in the UK and in the zonal variant (ZTCM) in particular (Turner, Bateman and Pearce, 1992). As far as the majority of UK valuation practioners are concerned ZTCM has almost reached the stage of complete rejection in favour of the 'individual' version (Garrod and Willis, 1991). A number of significant TCM problems, first raised in the UK literature in the early to mid 1970s, have not been satisfactorily addressed and continue to lie at the core of the critique of TCM:

a) First, there is what has become known as the 'endogenous price' problem. If potential visitors to a recreation site change their residence in order to be near to the site, the assumption that all zones have the same distribution of tastes collapses. If this type of behaviour does exist across a significant spectrum of the visitor population then the TCM (Clawson model) would substantially underestimate recreation site benefits. The price of a trip to the recreation site has become endogenous, and if corrections are not made then the estimated slope of the conventional travel cost demand curve will be too flat. The estimated consumer surplus for access value of the site will therefore be too small - see figure 4. Parsons (1991) has argued that endogeneity may be eliminated using an instrumental variables approach (e.g. place of work, job characteristics etc) in the questionnaire design and data analysis. Future TC surveys could include a question covering the importance of proximity to recreation sites in choosing place of residence. Split data sets could then be used to test for bias;

b) visitors may visit a site as only one of a number visited during a single trip. In this case it is not certain how much of an overestimation of benefits there would be;

c) some visitors may derive utility from the journey to a site ('meanderer' visitor). A complex behaviour pattern may be present again making it difficult to discern whether benefits are under or over-estimated;

d) the inclusion of a value for time spent travelling to a site while theoretically correct has nevertheless served to open a long debate on which empirical approach should be used.

During the late 1980s the TCM was applied in a study of coastal recreation in the UK (Green et al., 1990). Analysts sought to test the assumption that the value of enjoyment must be higher for visitors who travel further (incurring higher costs) to visit a particular site. Visitors to six seaside locations in England were questioned (using CVM) about their valuation of the enjoyment experienced on a day visit. The survey results were then compared with their travel costs. These costs were estimated by using a computer programme which computed the cheapest, quickest and shortest routes between the origin of the trip and the coastal site.

Results of this study did not support an unquestioning and extensive use of TCM in the UK. Only 2 sites (Dunwich and Spurn Head) produced data indicating a rising enjoyment value as travel cost and travel time and distance increased. For another site (Clacton) enjoyed value increased as trips were made from increasing distances, but only up to a threshold of about one hour's travel time. In the case of 2 further sites

(Frinton and Scarborough) the value of the day's enjoyment seemed to be negatively related to distance travelled. Finally, in the case of Filey, no association between the value of enjoyment and distance travelled was found.

Household production function (HPF) (including hedonic pricing) models also rely on individual actions in order to isolate features of their values. Empirical results for HPF and hedonic models applied to housing markets (and linked to air pollution or water quality) are not as extensive as that of TC models. US researchers believe that good quality hedonic pricing models are capable of providing credible evidence of a negative and significant relationship between air pollution and property value. Estimation of the representative household's marginal valuation of reductions in air pollution is less certain (Smith, 1992).

There is now a large body of research which elicits individuals' valuations of changes in some environmental resources using the CVM, in which individuals express their preferences by answering questions about hypothetical choices. CVM has been subject to criticism, particularly as a result of theoretical and experimental research by psychologists and economists into the problems of eliciting preferences. Supporters of CVM are currently attempting to address both reliability and validity questions.

A basic question for the implementation of the CVM is whether willingness to pay for benefits (WTP) or willingness to accept compensation for disbenefits (WTA) is the most appropriate indicator of value in a given situation. For cost-benefit analysis based on the Hicks-Kaldor compensation test, WTP would seem to be the appropriate measure for gainers from some resource-allocation decision, and WTA the proper measure for losers from that same reallocation. But as Harris and Brown (1992) have pointed out it is often not easy to conclusively identify gainers and losers since this judgement is itself influenced by the valuer's own perspective.

Willig (1976) claimed that WTP and WTA measures should, in the absence of strong income effects, produce estimates of monetary value that are fairly close (within 5%). However, since 1976 strong evidence has been accumulated which shows that for given environmental goods, WTA is significantly greater than WTP (40%+ divergence). In addition, WTA valuations seem to have greater variance than WTP ones, and are less accurate predictors of actual buying/selling decisions.

The format of the questions used to elicit valuations may be continuous (or 'open-ended'), i.e. asking respondents to state WTP or WTA without any prompts concerning possible answers, or discrete (or 'dichotomous'), i.e. presenting the respondent with a single buying price or selling price which must be accepted or rejected. Many intermediate formats are also possible, e.g. bidding games. These differences in format can produce systematically different responses (Desvousges, 1987; Loomis, 1990).

A number of explanations have been offered for the differences in valuations elicited by different formats:

i) There may be income effects, as predicted by Hicksian consumer theory. In a recent paper, Hanemann (1991) has argued that such effects could account for some observed WTP/WTA differences for public goods. He has calculated that a WTA measure five times greater than WTP can be justified in cases where the elasticity of substitution is low and/or the WTP/income rates is high, i.e. for unique, irreplaceable environmental assets which individuals care a great deal about.

ii) A psychological phenomenon loss aversion, may be important, especially in the case of potential losers in a resource change when WTAC questions are related to giving up things, rights or privileges (Schroder and Dwyer, 1988). As predicted by Tversky and Kahneman's (1991) theory of reference-dependent utility, valuations may be made relative to reference points, losses being weighted more heavily than gains. Such effects, which could account for some WTP/WTA differences, have been found experimentally (e.g. Knetsch and Sinden, 1984). Similarly, anchoring effects (or starting point bias) may cause differences between responses to discrete and continuous formats.

iii) WTA questions may be less readily understandable than WTP ones, since most people have more experience of buying goods, paying taxes, etc. than of selling. Similarly, continuous questions may be less readily understandable than discrete ones, since most people have more experience of choosing whether or not to pay stated prices than of stating valuations.

iv) The continuous format may have a stronger tendency than the discrete format to suggest opportunities for free riding.

v) Respondents may act strategically, i.e. making guesses about how their answers will be used and then give the answers that they believe will serve their interests best.

Overall, it is likely that merely identifying gainers and losers in some resource change situation will be insufficient to determine whether WTP or WTA is the most appropriate indicator of value. We need to know more about the motives of the valuer (Harris and Brown, 1992). Economics has much to learn from psychological research in this context. In fact, some of CVM's strongest critics are to be found outside the economics profession, in the ranks of the philosophers, psychologists, political scientists and scientists.

4.3 Debate Outside Economics

Sagoff (1988) has argued that economics makes a "category mistake" in its approach to environmental valuation. For him, it is not preferences but attitudes that determine people's environmental valuations. Thus people may not be willing to consider market-like transactions (assumed by CVM) involving public resources. CVM surveys pick this effect up in the form of refusals and "protest bids" (see figure 5). Some combination of individual preferences and public (collectively held) preferences will be held by any given individual who by necessity has to operate in daily life as both a consumer and a citizen. Thus the environment can be both a purchased commodity and a moral/ethical concern.

According to Sagoff environmental economics has no role to play in the determination of the goals of environmental policy. Environmental protection standards are determined by political, cultural and historical factors not by preference-based values. If economics has a role it is restricted to revealing the costs (social opportunity costs) of the pre-emptive environmental standards. But if action is taken on the basis of the opportunity cost analysis then an implicit valuation has been made. Nevertheless, from this viewpoint there is no role for direct monetary valuation (preference-based) of the benefits of environmental protection policy.

Other critics do not go as far as completely rejecting the validity of WTP/WTA measures of value, but instead argue that economic values are only partial values for many environmental resources. Thus Brennan (1992) seeks to distinguish so-called transformative value from the economic value of environmental assets. Environmental economists have gone a considerably way toward a taxonomy of economic values as they relate to natural environments (Pearce and Turner, 1990). The terminology is still not fully agreed, but the approach is based on the traditional explanation of how value occurs, that is, it is based on the interaction between a human subject (the valuer) and objects (things to be valued). Individuals have a number of held values which in turn result in objects being given various assigned values.

In order to arrive at a measure of total economic value, economists begin by distinguishing user values from non-user values. In a straight forward sense, user values derive from the actual use of the environment. Slightly more complex are values expressed through options to use the environment (option values). They are essentially expressions of preference (willingness to pay) for the preservation of an environment against some probability that the individual will make use of it at a later date. Provided the uncertainty concerning future use is an uncertainty relating to the "supply" of the environment, economic theory indicates that this option value is likely to be positive. A related form of value is bequest value, a willingness to pay to preserve the environment for the benefit of one's children and grandchildren.

Non-use values present more problems. They suggest values which are in the real nature of the thing but unassociated with actual use, or even the option to use the thing. Instead such values are taken to be entities that reflect people's preferences, but include concern for, sympathy with, and respect for the rights or welfare of nonhuman beings. Individuals may value the very existence of certain species or whole ecosystems. Total economic value is then made up of actual use value plus option value plus existence value.

Many environmental assets, according to Brennan, posses additional transformative properties such that 'exposure' to the assets causes a change in people's preferences. The impact of transformative values on people's preferences is said to be completely unpredictable and the degree of impact (when it occurs) will vary significantly from person to person. If natural things and systems possess transformative values, then it is argued they cannot be priced by economic analysis.

As far as we can see this transformative value argument can be accommodated within the total economic value approach. The transformative value is equivalent to a use or existence value, only it is latent, and for some individuals may never actually exist, i.e. their preferences are never changed (transformed) by contact with or knowledge of the specific thing that possesses the transformative property. At the policy level, if it is the case that natural things and systems possess transformative values then a conservation strategy based on the total economic value principle (in particular option and bequest values) would be sufficient to guarantee their future existence. So individuals may value (exhibit a WTP for) natural things and systems in order to retain the option to use them, or be transformed by them, some time in the future. Bequest value similarly expresses an individual's wish to retain options for their descendants.

Some scientists have argued that the full contribution of component species and processes to the aggregate life-support service provided by ecosystems has not been captured in economic values (Ehrlich and Ehrlich, 1992). There does seem to be a sense in which this scientific critique of the partial nature of economic values has some validity, not in relation to individual species and processes but in terms of the prior value of the aggregate ecosystem structure and its life-support capacity.

Since it is the case that the component parts of a system are contingent on the existence and functioning of the whole, then putting an aggregate value on wetlands and other ecosystems is rather more complicated a matter than has previously been supposed in the economics literature. Taking wetland ecosystems as an example, the total wetland is the source of **primary value** (PV) (Turner, 1988a; Turner, 1992). The existence of the wetland structure (all its components, their interrelationships and the interrelationship with the abiotic environmental) is prior to the function/service values. These latter values we term total secondary value (TSV) and they are conditional on the continued 'health' of the ecosystem. Each secondary value is dependent on the existence operation and maintenance of the multi-functional wetland system. The concept of total economic value (TEV) has two not just one limitation as previously supposed. TEV may fail to fully encapsulate the total secondary value (TSV) provided by an ecosystem, because in practice some of the functions and processes are difficult to analyse (scientifically) as well as to value in monetary terms. But in addition, TEV fails to capture the PV of ecosystems, indeed this "existence" or "glue" value notion is very difficult to measure in direct value terms since, it is a non-preference, but still instrumental, type of value.

We believe that this primary and secondary ecosystem value classification goes some way towards satisfying many scientists' concerns about the 'partial' nature of the conventional economic valuation approach (Ehrlich and Ehrlich, 1992). It is also a classification that avoids the instrumental versus non-instrumental value in nature debate which we believe has become rather sterile.

More formally[1]:

a wetland system provides a source or stock of primary value = PV = e ("existence" or

[1] Comments by Jason Shogrun and David Pearce on this value classification have been very helpful.

"glue" value of the ecosystem)

the existence of a 'healthy' wetland system (stable and resilient) provides a range of function and services (secondary values) = TSV;

Total Ecosystem Value (TV) > TSV

But, TSV = TEV (Total Economic Value)

total economic value (TEV) = total secondary value (TSV);

where, TEV = UV + NUV

and, UV = DUV + IUV + OV

NUV = EV + BV

TEV = (DUV + IUV + OV) + (EV + BV)

So, TV = (TEV, e)

TV = (TEV, 0) = 0

TV (O, e) $\geq$ 0

where e = "existence" or "glue" value of the ecosystem

UV = use value

DUV = direct use value

IUV = indirect use value

OV = option or potential use value

NUV = non-use value

EV = existence value

BV = bequest value

TV = total ecosystem value

What is clear is that the components of TEV (use and non-use values) cannot simply be aggregated. There are often trade-offs between different types of use value and between direct and indirect use values. Smith (1992) has also pointed out that the partioning of use and non-use values may be problematic, if it is the case that use values may well depend on the level of services attributed to non-use values. The TEV approach has to be used with care and with a full awareness of its limitations.

4.4 Towards an Understanding of the Valuation Process

Figure 6 summarises the various elements thought to comprise the full valuation process. The interaction between a person and an object (to be valued) involves perception of the objects and a process whereby relevant held values, beliefs and dispositions come to the forefront. Perception and beliefs are interrelated and together result in an unobservable sense of value (utility), which may then be expressed as an assigned value and certain behaviour (Brown and Slovic, 1988). Brown and Slovic conclude that the valuation context may affect how objects are perceived, the beliefs that become relevant, the utility experienced and the value assigned.

Information (existing and new) plays a key role in the valuation process. An individual's familiarity with the environmental commodity/context and the resulting perceptions are dependent on both the information stock and the provision of new information. The type and form of information supplied is particularly important in situations where direct perception is not possible and recourse to 'expert' knowledge is required.

Perceptions, information and beliefs all then feed into motivation. Harris and Brown (1992) identify what they call a responsibility motive in the environmental loss context. The motive is best represented as a spectrum of feelings extending from personal responsibility to a more general concern for the environment unrelated to use value. Randall (1987) has argued that all non-use values have their basis in the motive of altruism - interpersonal, intergenerational and Q-altruism (based on the knowledge that some asset Q itself benefits from being undisturbed). What this discussion of motivation does is to question the simplistic 'rational economic person' psychological assumptions that underpin conventional economic analysis. The motive of self-interest is only one of a number of human motivations and need not be the dominant one.

Maslovian psychology, for example, substitutes the concept of human needs for human wants and portrays needs in a hierarchical structure (Maslow, 1970) see Figure 7. Instead of an individual facing a flat plane of substitutable wants (as in conventional economics), Maslow conceives of the same individual attempting to satisfy levels of need. The satisfaction of higher level needs leads to a process of "self-actualisation". Self-actualised individuals would be expected to possess a strong responsibility motivation and hold non-use values. Such individuals might well be prepared to pay to maintain some environmental asset regardless of the benefits they themselves receive from that asset.

Because of the divergence between WTP and WTA valuations many practioners have taken the pragmatic decision to regard stated WTP valuations as reliable measures of true WTP and therefore to use CVM only in cases in which WTP is the appropriate measure of benefit. But then the question becomes what is the exact set of cases in which WTP is appropriate? Harris and Brown (1992) have argued that WTP is in fact the appropriate measure of welfare change for a number of situations. They identify only self-interested losers from a resource change as the appropriate group to be surveyed with WTA format questions. A mail survey undertaken (in 1988) by the same researchers indicated that 53% of the taxpayers' sample said that their state of

Idaho should pay for the loss of nongame wildlife with tax dollars, implying that all taxpayers should pay to cover this loss (a WTP rather than WTA approach). Only 32% of survey respondents said that only those responsible for the loss should pay to prevent it (WTA format).

Overall, there are grounds for cautious optimism about the role that economic valuation methods and techniques can plan in environmental resource management. While there will be a limit to meaningful monetary valuation of the environment, the precise nature of this limit is still an open research question, see figure 8. A large number of individual environmental asset valuation studies have now been completed. The next requirement, for policy making, is to develop classes of commodities and their benefit measures which can be consistently aggregated and disaggregated: and therefore to simultaneously distinguish environmental commodities/contexts in which monetary valuation is unlikely to provide valid/reliable value estimates.

4.5 Sustainability, Cultural Capital Depletion and Sustainable Livelihoods

Co-evolution is a local process, so local human subsystems are a significant starting point for the discussion of evolution in ecological economics (Berkes and Folke, 1992). Traditional ecological knowledge may be important and therefore cultural diversity and biological diversity may go hand-in-hand as prerequisites to long-term societal survival. Diverse cultures encompass not just diverse environmental adaptation methods and processes, but also a diversity of worldviews (technocentrism and ecocentrism) that support these adaptations. The conservation of this rapidly diminishing pool of social experience in adaptability (**cultural capital, K_c**) may be as pressing a problem of maintaining an environmental balance constitute as important a reservoir of information as the genetic information contained in species currently threatened with extinction (Perrings, Folke, Mäler, 1992).

The most convincing body of evidence for human self-organisational ability may be found in the literature of common property-resources (Ostrom, 1990). Institutions, co-evolution and traditional ecological knowledge are all components of the common property dilemma. There exist a large number of self-regulating regimes governing access to resources in common property and their scope in limiting the level of economic stress on particular ecological systems is clearly very wide (Bromley, 1992).

Any sustainable strategy for the future will have to confront the question of how a vastly greater number of people can gain at least a basic livelihood in a manner which can be sustained, many of the livelihoods will have to endure in environments which are fragile, marginal, and vulnerable (Chambers and Conway, 1992).

A working definition of sustainable livelihoods (SLs) would be: "a livelihood comprising the capabilities, assets (stores, resources, claims and access) and activities required for a means of living: a livelihood is sustainable which can cope with and recover from stress and shocks, maintain or enhance its capabilities and assets and provide sustainable livelihood opportunities for the next generation: and which contributes net benefits to other livelihoods at the local and global levels in the short and long term" (Chambers and Conway, 1992, p.27-8).

From the policy viewpoint the aim must be to promote sustainable livelihood security, via vulnerability reduction. Both public and private action is required for vulnerability reduction - public action to reduce external stress and shocks through, for example, flood protection, prevention and insurance measures; and private action by households which add to their portfolio of assets and repertoire of responses so that they can respond more effectively in terms of loss limitation. Chambers and Conway (1992) offer a range of monetary value and physical indicators for sustainable livelihood monitoring and assessment of trends (e.g. migration (and off-season opportunities), rights and access to resources, net asset position of households etc.).

5. SUSTAINABILITY AND ETHICS

For many commentators traditional ethical reasoning is faced with a number of challenges in the context of the sustainable development debate. Ecological economists would argue that the systems perspective demands an approach that privileges the requirements of the system above those of the individual. This will involve ethical judgements about the role and rights of present individual humans as against the system's survival and therefore the welfare of future generations. We have also already argued that the poverty focus of sustainability highlights the issue of intragenerational fairness and equity.

So 'concerns for others' is an important issue in the debate. Given that individuals are, to a greater or lesser extent, self-interested and greedy, sustainability analysts explore the extent to which such behaviour could be modified and how to achieve the modification (Turner, 1988b; Pearce, 1992). Some argue that a stewardship ethic (weak anthropocentrism, Norton, 1987) is sufficient for sustainability, i.e. people should be less greedy because other people (including the world's poor and future generations) matter and greed imposes costs on these other people. Bioethicists would argue that people should also be less greedy because other living things matter and greed imposes costs on these other non-human species and things. This would be stewardship on behalf of the plant itself (Gaianism) in various forms up to "deep ecology" (Naess, 1973; Turner, 1991).

The degree of intervention in the functioning of the economic system deemed necessary and sufficient for sustainable development also varies across the spectrum of viewpoints. Supporters of the steady-state economy (extensive intervention) would argue that at the core of the market system is the problem of "corrosive self-interest". Self-interest is seen as corroding the very moral context of community that is presupposed by the market. The market depends on a community that shares such values as honesty, freedom, initiative, thrift and other virtues whose authority is diminished by the positivistic individualistic philosophy of value (consumer sovereignty) of conventional economics. If all value derives only from the satisfaction of individual wants then there is nothing left over on the basis of which self-interested, individualistic want satisfaction can be restrained (Daly and Cobb, 1989).

Depletion of **moral capital (K_e)** may be more costly than the depletion of other components of the total capital stock (Hirsh, 1976). The market does not accumulate moral capital, it depletes it. Consequently, the market depends on the wide system (community) to regenerate K_e, just as much as it depends on the ecosystem for K_n.

Individual wants (preferences) have to be distinguished from needs. For humanistic and institutional economists, individuals do not face choices over a flat plane of substitutable wants, but a hierarchy of needs. This hierarchy of needs reflects a hierarchy of values which cannot be completely reduced to a single dimension (Swaney, 1987). Sustainability imperatives therefore represent high order needs and values.

6. CONCLUSIONS

The notion of sustainable development, despite the definitional ambiguities which surround it, has nevertheless proved to be a very useful device for clarifying a number of the underlying economic, political and ethical principles encompassed by the competing environmental ideologies and worldviews. In this paper we have defined four sustainability positions (on the basis of their economic, social and ethical foundations) which result in different policy prescriptions for the development versus conservation problem.

It seems to us that the strong sustainability approach has much to commend it. This variant of the sustainability paradigm can accommodate the constant capital assets rule, but also buttresses the rule (and its equity provisions) with a precautionary approach linked to the probable existence of some critical natural capital assets. Economic growth will need to be constrained in order to conserve these critical assets. The conservation case is strengthened because of the combined presence of uncertainty, irreversibility, non-sustainability and loss aversion in many real world environmental resource management contexts.

The very strong (zero growth) sustainability standpoint, however, is, we believe, not a necessary condition for the maintenance of an adequate quantity and quality of environmental goods and services. Because this standpoint is stultifying of development it carries with it a high risk of unacceptable social costs in terms of development benefits to the poorest members of the community now and in the future. Further, concern for the interests of non-human species does not require the adoption of any of the more extreme strong sustainability bio-ethical arguments. There is, we would argue, sufficient conservation potential in a sustainable use strategy based on anthropocentric (largely instrumental) values in nature, but constrained by the precautionary principle.

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