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## A pluralistic approach to ecosystem services evaluation

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**A Pluralistic Approach to Ecosystem Services  
Evaluation**

By  
Kerry Turner

*CSERGE Working Paper EDM 10-07*

# **A Pluralistic Approach to Ecosystem Services Evaluation**

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## **Introduction**

The purpose of this note is to contribute to the emerging common understanding between the natural science, economic and social science perspectives on the 'value' of ecosystems and biodiversity. Given the inherent complexity of nature it is not surprising that the concept of 'value' is capable of multiple interpretation and meaning. A number of different dimensions of nature-based value can be simply discerned and evaluated in different ways eg in monetary terms via economic analysis; in biophysical and geochemical terms via natural science; and in more qualitative terms via sociology, cultural geography, arts and humanities etc. Each of these value dimensions has validity in its own domain.

Environmental philosophers have, however, constructed a generic value typology with four categories: anthropocentric instrumental value, which maps closely on to the economic concepts of use and most of non-use values; anthropocentric intrinsic value, a culturally dependant concept which is linked to human stewardship of nature motivation and which requires a human valuer to ascribe intrinsic value to non-human nature; the economist's idea of existence value can overlap into this value category. The other two value categories - non-anthropocentric instrumental value and non-anthropocentric intrinsic value are less directly relevant to the NEA initiative, unless in the latter value category's case a radical ethical position is accepted as the societal norm, which is currently not the case (Hargrove 1992).

Existence value derives from individuals who feel a benefit from just knowing that, for example, an ecosystem and/or its component parts, does exist and will continue to exist somewhere on the planet. The economic valuation literature has yet to reach a

comprehensive consensus on whether use and non-use value can be formally distinguished using standard welfare economic measures. The use of survey-based methods such as a contingent valuation and choice experiments to elicit monetary expressions of existence values is still open to debate on the grounds of validity and reliability. The conventional economic assumptions about human motivations and behaviour can be seen as quite restrictive and findings from the behavioural economics and psychology literature have served to 'enrich' our still inadequate understanding of cognitive behaviour. It seems that so-called bequest motivations (i.e. the requirement to pass on over generational time an 'environment' which can yield at least a constant set of 'opportunities') existence value motivations and altruistic motives may all be relevant and real in certain environmental loss contexts.

Analysts disagree over how to interpret this set of possibly overlapping motivations and behaviours. Some see the welfare effect as an individualised 'warm glow' effect connected to the act of giving, while others insist that 'pure altruism' is required for existence value and can be recognised. The debate is also further complicated by consumer-citizen distinctions that can be made. In the latter role, individuals may hold social preference values and motivations which may be best elicited through group discussion/involvement. If one accepts the position that only individual values are the 'real' values that should be taken into account in the policy process and that individual behaviour is dominated by self-interest and self-regarding motives, then only a restricted version of existence value (contaminated by the 'warm glow' effect) is possible. Thus value estimates derived from contingent valuation surveys will not necessarily indicate 'true' economic value derived from public goals such as the ecosystem service gain/loss under test. If, on the other hand, one is persuaded that citizen-type motivations and behaviour can be recognised, then other regarding motives and social preferences 'true altruism' exist. Group-based focus gatherings or other 'deliberative' processes may then offer a mechanism through which bequest and existence values can be elicited.

To summarise, focusing on just anthropocentric instrumental and intrinsic value in nature, it is important to note that the former value concept is usually interpreted in economic analysis in terms of an individual person (or sometimes aggregated household) and their preferences and motivations. The latter value concept however can also be viewed in a collectivist way, motivations and preferences which can be assigned to groups and culturally transmitted and assimilated over time as social norms. These cultural values may not be capable of meaningful and full monetary expression, but nevertheless they significantly signal that human well-being and quality of life is a function of both individual wants satisfaction and the fulfilment of a variety of social and health related and cultural collective needs. Cultural values are shared experiences fostered by and within 'groups' often over long periods of time and often connected to specific local places and landscapes.

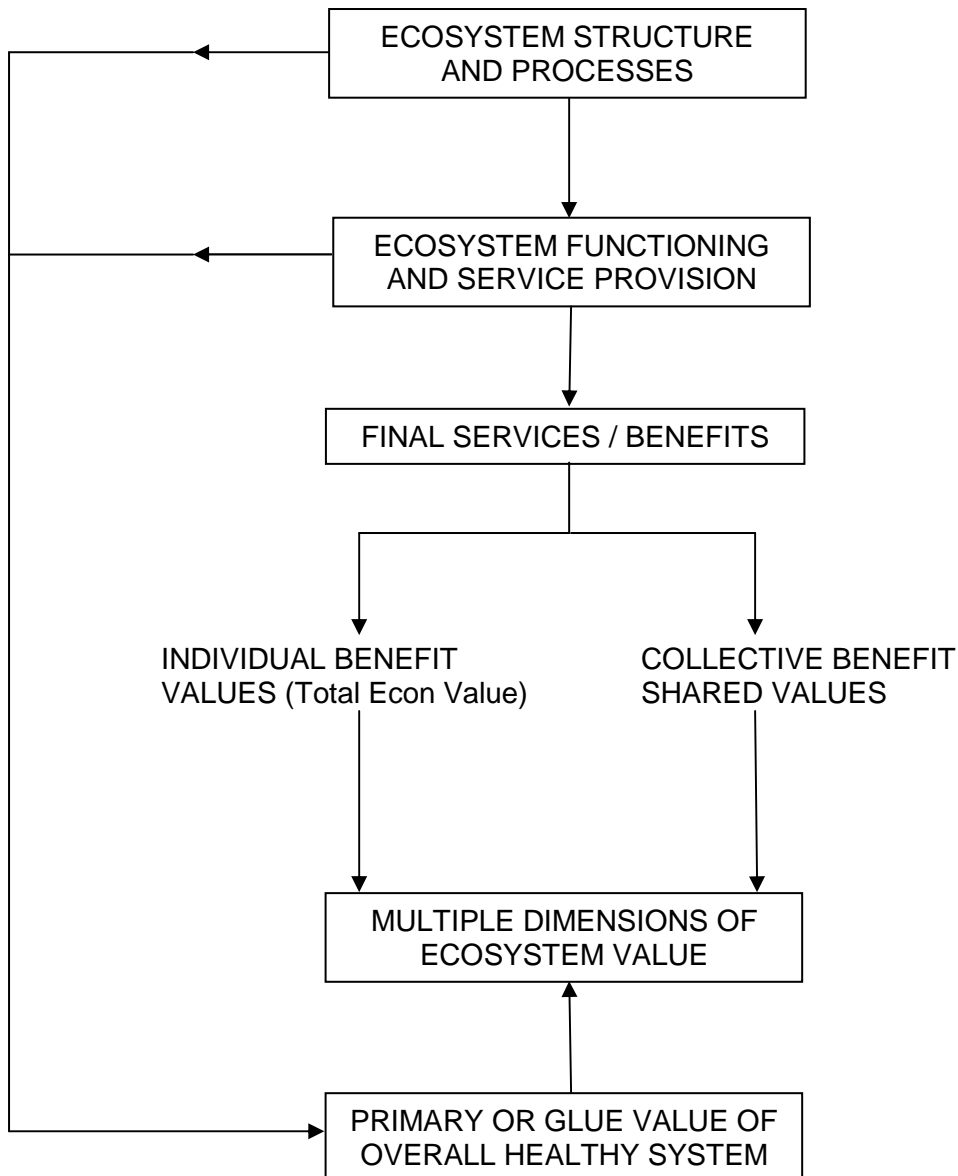
### **Environmental System Value**

A concern raised by some social science and natural science communities is that ecosystem services classifications and approaches (such as MEA & NEA) carry with them an inherent danger that leads to complete 'commodification' of ecosystems and a consequent policy and management failure. The failure manifests itself in terms of an over-concentration on those ecosystem services and benefits of direct and indirect use / non use to humans, with the risk of over exploitation and system change or collapse. The NEA conceptual framework recognises that it may be the case that when the value of whole environmental systems are concerned, conventional economic valuation ( restricted to the flow of service benefits) may not be sufficient. The framework accepts the need to assess and conserve the structural and process / functional value of 'healthy' evolving ecosystems, despite the formidable uncertainties surrounding likely thresholds / tipping points for system change. The fundamental life-support services, labelled 'intermediate services' in the conceptual framework, clearly are valuable and the focus on the flow of assigned ecosystem benefits values is not meant to deny this.

Healthy ecosystems, anchored to a sufficient configuration of structure and process, have 'prior' value (labelled primary, glue or infrastructure value) in the sense that the continued existence of the system 'integrity' determines the flow of all the instrumental and intrinsic values related to final ecosystem services and benefits. So total system value is always greater than total economic value (Gren et al 1994; Turner 1999).

Figure 1 summarises the arguments presented so far.

**Fig 1 SIMPLIFIED ECOSYSTEM VALUES TYPOLOGY**



Given the policy goal of sustainable development, it has been argued that rather than an income flow-based approach to ecosystem valuation, a stock-based approach is more appropriate. Such a stock-based approach could be based on the measurement concept of 'inclusive wealth' (Arrow et al 2003; Dasgupta and Maler, 2001). Typically, the core sustainable development resource allocation constraint rule has been interpreted as, that the present value (ie discounted) of future well being (utility) linked to consumption flows must be maintained over time. But an equivalent rule can be derived in terms of the 'inclusive wealth' concept and the value of natural and other capital stocks at a given time. Value is defined as the quantity of current stocks multiplied by their shadow prices, and 'inclusive wealth' is maintained if the value of capital stocks is constant over time. The shadow price of a capital asset today is the present value of the change in utility that would be caused by a marginal change in the quantity of the capital asset today .

However, shadow prices must be fully inclusive ie they must reflect all relevant costs and benefits to current and future generations. A similar proviso is also required in the income flow value-based approach and both approaches are therefore limited by the lack of perfect information ( natural science , social and economic). Norgaard (2009) has provided a critique of the stock-flow ecosystem services approach in which, among other issues, he focuses on the state of scientific/ecological knowledge. He concludes that science is far from being able to predict smaller shifts in the delivery of ecosystem services and that there is no scientific consensus around threshold effects which can cause ecosystems to shift from one state to another and /or suffer stock collapse. So both risk and uncertainty problems are present when ecosystem services are exploited for their human related benefits flows. The maintenance of ecosystem integrity ( conditioned by some poorly understood minimum configuration of ecosystem structure and process) can help to maintain resilience capacity ie the ability of the ecosystem stock to withstand stress and shock without significant state

change. The resilience capacity has an economic value (price) but so far efforts to empirically estimate it have been rare and preliminary (Maler et al 2008).

In the light of the prevailing uncertainty surrounding ecosystem threshold effects and the risks and consequences of ecosystem collapse, a number of pragmatic management responses have been suggested eg the imposition of safe minimum standards etc. These mitigation measures are reviewed later in the next section, but Norgaard (2009) concludes that only major institutional change can provide an adequate response to current unsustainable consumption patterns across the globe. Thus the ecosystem services approach can only be a part of a larger solution.

### **Irreversibility and Related Concepts**

A number of issues surround the notion of irreversibility and related concepts of threshold effects and tipping points. While all these terms are now in use in the environmental conservation and economics literature, we find it helpful to refer to thresholds in the context of individual ecosystems or landscape ecology limited to the regional spatial scale. This reserves the term, tipping points, to describe global scale system and subsystem non linear and abrupt reactions to environmental change pressures (Rockstrom et al 2009).

Ecosystems function via feedbacks between different components of structure and process. When the feedback effects are positive any given initial perturbation (stress or shock) of the system will be amplified, when a positive feedback occurs the prevailing state of the system may be such that a complete switch into a different state is triggered (a classic case is the enrichment of shallow lakes via excessive N & P inputs from the surrounding catchment, causing abrupt change in water quality and aquatic plant and fish etc communities). The initial ecosystem state prior to the 'flip' is then a threshold or a bifurcation. The capacity of an ecosystem to 'absorb' stress or shock and remain in its prevailing state is known as resilience. It is still a matter of scientific debate whether greater diversity in ecosystems provides a buffering capacity (greater stability or resilience) and in which specific contexts



(Tisdell, 2009 and Worm et al 2006). A further degree of uncertainty surrounds the question of whether the ecosystem state change is reversible or irreversible in the future. This is a far from straightforward question and we currently lack sufficient scientific and other knowledge to be able to offer robust prescriptions. In the shallow lake example cited earlier, it is the case that remedial management actions (such as sediment pumping, N & P abatement etc) can restore water quality & other losses. But even in this case, the timing and extent of the necessary abatement programme is not clear cut, with adverse consequences for the overall costs of action. In more complex contexts, irreversibility is even more difficult to pin down, either because of current technological / scientific data and means deficiencies, or impracticability constraints in the form of significant cost burdens and governance limitations. Future scientific and technological breakthroughs and socio-economic conditions and preferences may or may not lead to less constraints, but in any case they are currently only predictions or complete unknowns.

Given that information about ecosystem functioning and dynamics under contemporary environmental change conditions (local through to globalisation) is incomplete, there is a positive probability that a given ecosystem in a given change context will be pressurised into a thresholds zone and across a point, causing it to flip to a less desirable new state. The probability of flipping is lower as resilience is maintained / increased and management interventions to 'conserve' resilience are therefore important. Resilience capacity can be regarded as capital stock (natural capital) which yields an insurance service and benefit (Maler et al 2007). As a stock, in principle it has an accounting price, defined as, the change in the expected change in net present value of the expected future ecosystem services resulting from a marginal change to resilience today (Maler et al 2007). In practice, data (time series and other) constraints have so far precluded the monetary valuation of this insurance service, with the one exception of an agroecosystem study in Australia.

Two different approaches have been put forward in the environmental economics literature as coping strategies for the irreversibility problem. The first was a modified CBA method, known as the Krutilla-Fisher (K-F) approach. It laid down that in relevant preservation versus development situations, the benefit of the preservation option should be factored into the CBA equation. Preservation benefits forgone should be treated as part of the costs of development and should be assumed to increase through time because of the relative price effect. The development benefits should have an offsetting discount factor, in addition to the 'basic' discount rate because of 'technological obsolescence'. It is also the case that the present value of development can be very sensitive to the preservation relative price effect and the obsolescence factor (Pearce & Turner 1990). Given the prevailing information gaps, it is recommended that the benefit of the doubt be given to preservation over development in all cases where benefits and costs are reasonable closely balanced. The K-F strategy was designed to cope with ecosystem and landscape asset losses characterised by uniqueness and national / international significance. The dilemma was how to cope with possible irreversible losses of unique assets such as designated national park lands. But it may be the case that some potential ecosystem losses are of 'local' significance and 'uniqueness'. In these cases a conservation versus development trade-off needs to be addressed.

In these 'local irreversibility' situations a 'shadow project' approach in which sustainability considerations are integrated into the CBA calculus may be relevant. The decision maker is asked to consider a range of decisions about development options and impose a sustainability constraint into the decision support system and process (ie to keep the stock of natural capital ( $K_n$ ) constant over time by suitable compensatory expenditures). The sum of the ecosystem damage done by a whole sequence of development projects would have to be offset by separate projects within the 'portfolio' of decisions being made. These compensatory projects would not have to pass the positive B-C ratio test (Barbier, Markandya & Pearce, 1990). The precise form of 'acceptable' compensation will vary from context to context and is an under-researched area. Roach & Wade (2006) for example

have examined the use of so-called habitat equivalency analysis which estimates ecological service loss and then scales restorative ecological compensation to offset the damage impact.

A second line of argument if irreversibility concerns are relevant incorporates the notions of the precautionary principle and the safe minimum standard. There is a line of reasoning that can link ecosystem diversity and resilience maintenance (with 'primary' / 'glue' / 'infrastructure' values in nature, alongside the 'insurance' value noted earlier) together with a support for the precautionary principle and strong sustainability (Gren et al 1994, Turner et al 2003). The precautionary principle is itself shrouded in ambiguity and CBA can provide a useful filter for it if a 'safe minimum standards' (SMS) interpretation covering species, habitats and ecosystems is accepted (Crowards, 1998). This goes back to the work of Ciriacy-Wantrup (1952) and Bishop (1978) in which it was advocated that a project should be rejected if irreversible losses of nature could consequently occur, unless the social costs of doing so were prohibitive (a modified minimax rule). Thus decision makers facing the prospect of very high preservation or conservation costs might choose to sanction a development option even though it carries a small risk of significant ecosystem damages. Nevertheless, judging what is or is not an "unacceptable large" or "tolerably low" social cost can be informed by ecology, economics, risk analysis etc, but ultimately is a 'political' call. Ethical and political choices will have to be made and deliberatively agreed. Over time, the aim should be to improve our understanding of ecosystem functioning so we can move towards more situations where we are dealing with risk rather than uncertainty.

It has recently been pointed out that, at the global spatial scale, many subsystems of Earth are sensitive to threshold effects and if a 'tipping point' is crossed unacceptable environmental change could be triggered (Rockstrom, 2009). The claim is that most of the thresholds can be defined by a critical value for one or more control variables. Nine processes have been identified (climate change; rate of biodiversity loss; interference with

the nitrogen and phosphorus cycles; stratospheric ozone depletion; ocean acidification; global freshwater use; change in land use; chemical pollution and atmospheric aerosol loading) the first three of which have already reached the threshold zone.

Rockstrom et al adopt a precautionary safe minimum standards perspective and propose that the planet must be kept in a “safe operating space” through the observance of quantified boundaries. In the case of biodiversity loss they advocate a boundary of ten times the background rates of extinction. Because of the many gaps in our knowledge this boundary should be considered as preliminary, but it seems clear that the current rate of species loss (100 to 1,000 times more than what could be considered natural) will lead to significant reductions in ecosystem resilience. A further concern is that globalisation has resulted in a rate and extent of economic activity sufficient to pressurise a range of earth processes simultaneously. This means that the planetary tipping point boundaries are tightly coupled and piecemeal abatement strategies are unlikely to be sufficiently effective.

The implementation of this global safe minimum standards strategy will be controversial and will require concerted and targeted science and social science research efforts to underpin it. But most of all it requires a radical overhaul of the governance processes controlling international trade and finance and resource exploitation etc. In the interim, recent work by Lenton et al (2008) has proposed the use of early warning systems which identify systems that are likely to cross ‘tipping points’ and are relevant to policy and accessed by humans (“tipping elements”). Historical data and predictive modelling (eg degenerative fingerprinting) may then be used to locate tipping points.

### **Cultural Ecosystem Service Values and Human Well Being.**

Human well being ( or any given individual’s perception of well being ) is a function of a set of social, psychological, economic and cultural factors and underpinning environmental conditions. While mainstream economics has tended to concentrate on the individual as an

isolated being with a behaviour dominated by self-interest and material want satisfaction, insights from psychology and behavioural studies have served to counterbalance this perspective. They emphasise the individual as a member of groups, neighbourhoods, communities etc (Maslow 1970; Hirsch 1977; Layard 2005 ). In this context individual behaviour and perceptions of well being are then heavily influenced by group attitudes and actions. Any index of psychological well being will contain personal development needs such as good health, autonomy and purpose in life, positive social relationships and self acceptance and spiritual and religious goal satisfaction. Personal development needs satisfaction are enabled , among other things , through the provision of basic want satisfaction ( food, shelter etc ), personal security and a stable and fair society with equal opportunities.

But a key enabling condition is also a healthy environment providing for a secure and reliable flow of ecosystem services, including valuable social and cultural services. The psychological well being index is fundamentally conditioned by the interactions between humans and the natural environment. Both mental and physical health are positively affected by contact with and access to 'green spaces' ( in local neighbourhoods and other places, landscapes etc ). Young peoples' educational experiences can be enriched by environmental exposures, both passive and active. A wide range of outdoor leisure and recreation activities provide important well being effects through the satisfaction of a variety of wants/needs, including culture, art and media appreciation.

The evaluation of these contributions to overall human well being provided by ecosystems will need to encompass both the monetary and other equally valid dimensions of 'value'. The direct use , by individuals, of outdoor leisure, recreation, cultural heritage and local green spaces 'for example' can often be valued via willingness to pay for visits measures. The beneficial mental and physical health effects provided for through contact with the natural environment can also be valued in monetary terms and via health related quality of life indicators. But cultural services pose a more complex evaluation problem.

The evaluation of cultural heritage (broadly defined) must be linked to the discourse of culture, memory and language, playing out over long periods of time. The embedded intrinsic values are 'shared' values, formed and altered through groupings of people and their collective will and sense of identity. The important role of regulatory and other governance institutions ( e.g National Park Authorities etc, etc ) in cultural services provision , protection and interpretation must also be recognised. The values transmission mechanisms are, among others , art, literature and media. So the 'value' of local places and landscapes, for example, cannot be adequately captured just in terms of individual visit rates and willingness to pay. This approach alone misses the more collectively based intrinsic value of cultural identity, heritage and bequest capability.

A new set of deliberative approaches to valuation have emerged which combine stated preference methods with the discourse-based techniques of political science (Spash, 2008). These methods involve small groups usually of between five to 20 individuals, who are selected to represent 'society' and be provided with information about ecosystem service benefits, which they discuss and formally deliberate in transparent a process as possible. By engaging in group discussion, individuals are believed to be exposed to a much richer set of information, attitudes and experiences, enhancing understanding of the pertinent issues. The deliberative component is explicitly introduced on the belief that participant individuals will explore motivations and ethnics beyond their own personal welfare. Value results can be expressed in individual or group willingness to pay terms, or more qualitatively. Protocols covering group discussion procedures are also being refined (Wilson and Howard, 2002); and the analysis of group dynamics goes back to after the 1970's and the Symlog (System for the Multiple Level Observance of Groups) methodology (Bales and Cohen,1979).

### **Happiness data, human well-being and ecosystem services**

With its origins in the 1970's (Easterlin 1977) scientific interest in the so-called 'happiness approach' and empirical measures of human happiness increased significantly in the last

decade (Layard 2010; Oswald and Wu 2010). This emerging field of research offers a novel way of valuing environmental goods which model individuals' self-rated happiness as a fraction of their incomes and the prevailing environmental conditions (Welsch and Kuhling 2008). The approach seeks to measure individual stated subjective well-being (happiness or life satisfaction) through large scale survey data collected on a regular basis. The data on subjective well-being has been used in economic research as an empirical approximation for "experienced utility", which is an ex post hedonic quality (satisfaction) associated with an act of choice (Kahneman et al 1997). Some surveys use verbal categories, while others use a numerical scale (eg 1 = dissatisfied to 10 = satisfied). Measures of happiness of life satisfaction correlate well with each other and according to factor analysis represent a single unitary construct (Welsch, 2010). A mix of personal and demographic characteristics as well as socio-economic factors are significantly correlated with happiness. Both the degree of urbanisation and the provision of environmental amenities, for example, are significant variables (negative and positive respectively). So using happiness data for environmental valuation we may be able to estimate the value of ecosystem service benefits in a different way to standard cost-benefit analysis and willingness to pay methods. By correlating people's reported subjective well-being with ecosystem service benefits and personal income (including lagged individual income and other people's income) it is possible to identify the environmental quality and income utility traction and to estimate the implied utility-constant trade-off between them (eg the increase in income necessary to compensate individuals for any decline in environmental benefit provision). So far air pollution, water pollution, noise, climate conditions and natural hazards have been valued using the happiness approach. To the extent that ecosystem functioning contributes to the maintenance/improvement of air and water quality etc, valuation of ecosystem service benefits is possible via this 'happiness approach'.

## References

- Arrow K,J, Dasgupta P and Maler K-G (2003) Evaluating projects and assessing sustainable development in imperfect economies, *Environmental and resource Economics* 26:647-685.
- Bales R,F and Cohen S,P (1979) *SYMLOG: A system for the multiple level observance of groups*, Free Press, N York.
- E.B. Barbier et al (1990) Environmental sustainability and cost-benefit analysis, *Environment and Planning A* 22: 1259-1266.
- R. Bishop (1978) Endangered species and uncertainty: the economics of a safe minimum standard, *American Journal of Agricultural Economics* 60: 10-18.
- S.Ciracy-Wantrup (1952) *Resource Conservation: Economics and Policies*, University of California Press, Berkeley.
- T.Crowards (1998) Safe minimum standards: costs and opportunities, *Ecological Economics* 25: 303-314.
- Dasgupta P and Maler K-G (2001) wealth as a criterion for sustainable development, *World Economy* 2:19-44.
- Easterlin R (1974) Does economic growth improve the human lot? Some empirical evidence. In P A David and M W Reder etc *Nations and Households in Economic Growth: Essays in Honour of Moses Abramovits*, NY. Academic Press 89-125.
- Gren, I-M et al (1994) Primary and secondary values of wetland ecosystems, *Environmental and Resource Economics*,4:55-74;
- Hargrove,C. (1992) Weak anthropocentric intrinsic value, *The Monist* 75:183-207.
- Hirsch, F. (1977) *Social Limits to Growth*, Routledge and Kegan Paul, London.
- Kahneman D, Walker P and Sarin R (1997) Back to Bentham Exploration of experienced utility, *Quarterly Journal of Economics* 112: 373-405.
- Layard,R.(2005) *Happiness :Lessons from a New Science*, Penguin Books, London.
- T. Lenton et al (2008) Tipping elements in The Earth's climate system *PNAS* 105: 1786-1793.



K.G. Maler et al (2007) Accounting for ecosystem services as a way to understand the requirements for sustainable development, PNAS 105: 9501-9506.

Maslow, A. (1970) Motivation and Personality, Harper and Row, New York.

Norgaard R. (2009) Ecosystem services: From eye-opening metaphor to complexity blinder, Ecological Economics doi:10.1016/j.ecolecon.2009.11.009.

D.W. Pearce & R.K. Turner (1990) Economics of Natural Reserves and The Environment, Harvester Wheatsheaf, Hemel Hempstead.

B. Roach & W Wade (2006) Policy evaluation of natural resource injuries, using habitat equivalency analysis, Ecological Economics 58: 421-433.

J. Rockstrom et al (2009) A safe operating space for humanity, Nature 461: 472-475.

Spash C,C (2008) Deliberative monetary valuation and the evidence for a new value theory, Land Economics 84: 469-488.

Turner, R.K.(1999) The place of economic values in environmental valuation. In I.J. Bateman and K.G. Willis eds Valuing Environmental Preferences, Oxford University Press, Oxford.

R.K. Turner et al (2003) Valuing nature: lessons learned and future research directions, Ecological Economics 46: 492-510.

Welsh H and Kuhling J (2008) Using happiness data for environmental valuation: issues and applications, Journal of Economic Surveys 23: 385-406.

Welsh, H (2009) Implications of happiness research for environmental economics, Ecological Economics 68: 2735-2742.

Wilson M,A and Howarth R,B (2002) Discourse-based valuation of ecosystem services: establishing fair outcomes through group deliberation, Ecological Economics 41: 431-443.