THEORIES OF FAILURE, FAILURE OF THEORIES AND NON-MARKET VALUATION: A SURVEY

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Abstract

Non-market valuation of environmental benefits and costs plays a crucial role in environmental decision-making both at the micro and at the macro-levels. This paper basically reviews three theoretical developments embedded in economics, which can provide frameworks to value non-market benefits and costs of changes in environment. The major conclusion of this paper is that non-market valuation should be treated as 'derived demand' from any of these theoretical frameworks so that appropriate environmental policy measures can be initiated in the relevant areas.

Introduction

The central focus of environmental economics is how to value non-market environmental benefits and costs in monetary terms (Markandya, 1998). This concern is based on the fact that many of the environmental goods and services are either 'non-marketed' in nature or traded in imperfect markets, eventually leading to inefficient resource allocation pattern and, therefore, their true opportunity cost has to be estimated properly for the re-allocation to be Pareto optimal. The need for such an allocation arises from the fact that not only environmental degradation but also any policy decision to reverse it usually results in a 'trade-off' in welfare between different users.

Non-market valuation assumes paramount importance in environmental and natural resources management in developing countries because, 'many of the environmental assets that people generally feel are very important are in the developing world' (Pearce, 1993:15). Moreover, changes in the flow of non-market benefits derived from these environmental assets affect the basic livelihood of a considerable number of economic agents in these countries (see Dasgupta, 1996; Warford, 1989: Jodha, 1986). Therefore, to understand the extent of the changes in the resource flow and their marginal impact on the consumer or producer surplus is the ultimate objective of improving the social welfare. Hence, transforming the trade-off situation (a zero-sum outcome) into a 'win-win' situation (a non-zero sum outcome) warrants estimation of either

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the Hicksian *compensating variation or the equivalent variation*[1] (Freeman, III 1993) resulting from un-priced environmental goods and services entering into the production and utility functions of the individuals.

During the last few decades, the literature on non-market valuation has been strengthened by a large number of theoretical and empirical studies, including a good amount of empirical studies pertaining to developing country issues.[2] One of the important conclusions emerging from this literature is that the demand for non-market valuation, enriched both by theoretical and empirical studies, is derived from two major theoretical developments in environmental economics, namely, the *theory of market failure* and the *theory of government failure*. In the first instance, the *theory of market failure*, grounded in the Pigouvian externality analysis, has strengthened the microeconomic foundation of environmental economics mainly through the development of social cost-benefit analysis. On the other hand, the *theory of policy (or government) failure,* which originated recently, has laid the foundation for a macroeconomic analysis of environmental issues such as correcting for environmentally distorting macroeconomic policies, formulation of environment-friendly policies, estimation and monitoring of sustainable income, etc. This theory has also been strengthened by a group of environmental economists who argue that *policy failure* as such can be explained by the *failure* of modern macroeconomic theories to treat 'environmental or natural capital' (Costanza and Daly, 1992) as one of the integral and essential parts in their analyses. This school of thought, embedded in the Hicksian income analysis, not only suggests revision of the macroeconomic theories to incorporate natural capital as a constraining factor of economic growth but also prescribes certain practical policy measures by which the macroeconomic indicators such as national income can better reflect changes in the natural capital stock enhancing the sustainability of the national income.

All the above developments ultimately lead to placing importance on non-market valuation. In the case of market failure, for instance, the economic values of non-market benefits and costs of environmental actions have to be necessarily incorporated in the extended cost-benefit analysis so that the externalities can be internalised. In the case of policy failure, economic valuation is required in order to understand the extent to which environmental changes occur due to various alternative macroeconomic policies. Incorporating environmental capital into the macroeconomic models again helps place importance on economic valuation so that the environmental issues can be adequately incorporated into the model.

The major task of this paper is to systematically review the evolution of the theories of *failure* and *failure of theories* discussed above and how these developments lead to 'derived demand' for non-market valuation. In the process, we argue that *non-market valuation of environmental*

benefits and costs should fall under any one of these theoretical frameworks so that appropriate environmental policy measures can be initiated in the relevant areas. As a prelude to such an exercise, we first discuss the theory of market failure and its contribution to the development of microeconomics of environment especially in the form of cost-benefit analysis (CBA) in environmental economics literature, which, in turn, lead to 'derived demand' for non-market valuation.

Theory of Market Failure and Cost-Benefit Analysis

The theory of market failure emerged in the field of economics after Marshall (1920) introduced the concept 'externality', which was later strengthened by Pigou (1938).[3] While Marshall's (1920) analysis has been restricted only to the positive externality (see Barbier, 1993), the Pigouvian analysis deals extensively with the 'negative' externality problem, which has become a part of microeconomic theory (Cropper and Oates, 1992). The major focus of the Pigouvian negative externality analysis was mainly to address the problem of market failure that caused divergence between the 'social and private net product' (Pigou, 1938). Since there was a need for internalising the externality to achieve the social optimum in rescurce allocation, the standard policy prescription in Pigouvian analysis was that the government should intervene in those activities where the market has failed to bridge the gap between the social and private net products. In other words, it was argued that the price of the polluting commodity does not reflect that part of social cost imposed on the third party and, therefore, the government should impose either a tax or an effluent fee on per unit of emission from those polluting activities to internalise the negative externality. The level of tax or fee should be determined at that point where the marginal social cost of controlling pollution is equal to the marginal social benefits restored (see Cropper and Oates, 1992). As a basic requirement, the Pigouvian solution warrants the social cost-benefit analysis of an environmental policy intervention of the government.

As a normative tool, CBA suggests that for government intervention to be economically and socially efficient, the total discounted social benefits of that intervention should exceed the total discounted social costs, or alternatively, the discounted net benefit arising out of the intervention should be positive (Hanley, 1999). In a more rigorous neoclassical economics sense, the marginal benefits and the marginal costs of the pollution control policy of the government should be equated at its optimum so as to achieve Pareto efficiency. In principle, the estimation of marginal social benefits is a necessary condition for determining the optimum level of tax or fee for the appropriate policy intervention, but in practice estimating the social benefits is a more difficult task (Cropper and Oates, 1992).

Since there is no such 'optimum level' of pollution tax or fee that can be encountered in the real world, the Pigouvian solution for market

failure remains a powerful tool only at the theoretical level. Having understood the difficulty of estimating the social costs and benefits to appropriately deal with market failure, it was suggested alternatively that a 'second best solution' should be possible in which politically arrived at pollution standards can be set and these standards should be achieved at *least-cost*[4] still being within the Pigouvian effluent tax framework (Baumol and Oates, 1988; Burrows, 1979). This least-cost solution basically suggests that the level of tax should be equal to the marginal cost of controlling the effluents at the prescribed level so that polluters can have incentives to control pollution rather than paying the tax.[5] The underlying assumption here is that the politically arrived standard-setting which is based on scientific information is far from any imperfection and, therefore, the only task of the economists is to find out the marginal abatement cost (MAC) of controlling the pollutants up to their prescribed standards and fix the effluent tax at that point where the MAC is equal to the least-cost of achieving these standards. However, the second best solution failed to take into account the fact that the standard-setting is based only on scientific information and not on the changes in economic welfare due to the impact of the remaining level of pollution on production and utility. Hence, the second best solution, in practice, may still leave the divergence between private and social costs intact, leading to demand for non-market valuation.

Despite the difficulties, the second-best solution, continues to dominate the pollution control policy regime in many of the countries, thanks to the non-existence of an effective alternative solution. It should be noted that in almost all countries, the standards are complemented with regulation, charges and tradable permits. This is because deriving the marginal social benefit curve within the Pigouvian framework is always constrained by the lack of information on the benefits. Even though the least-cost solution is considered an effective outcome in view of the difficulties in estimating the Pigouvian type social benefits and social costs, achieving the first-best, i.e. the Pigouvian solution, is always the necessary condition for correcting market failure, given the fact that the opportunity cost of resource transferred to pollution control activities in developing countries is considerably high. Moreover, the following caveats still justify attaining of the first-best solution in the existing pollution control regime: (i) whereas estimating the private cost is relatively easy, a similar exercise in the case of social cost still poses difficulty. Moreover, the concept 'least-cost' includes not only the private cost of controlling pollution but also the social cost in terms of welfare foregone due to reduction in output, etc., which needs to be quantified in money terms (Cropper and Oates, 1992); and (ii) many empirical studies have shown that the 'actual pollution control costs' at the industry level are several times greater than the least-cost levels (Tietenberg, 1985). The underlying implication of this phenomenon is that the MAC based on the least-cost criterion may exceed the benefits restored and, therefore, this creates a strong case for the

ultimate goal of quantification of the benefits restored for a useful comparison.

The theory of property rights emerged as a powerful alternative to the Pigouvian-type government intervention solution to explain market failure. The property rights theory views market failure as arising purely out of the existence of ill-defined individual property rights over the resource domain and, therefore, it basically suggests that assigning property rights, in the absence of transaction costs, can always lead to achieving Pareto efficiency in environmental resource allocation (Coase, 1960). This solution, however, is found to be less practicable when the following conditionalities do not hold:(i) existence of less and identifiable number of polluters and pollutees in the economy; (ii) pollutants are homogenous in nature; and (iii) absence of transaction costs. Though this theory can be the best alternative to the Pigouvian approach provided all the above conditionalities remain valid, estimating the social benefits/ costs for determining the price of the tradable permits (or rights to pollute), etc., does *not* yet reduce the burden of the non-market valuation practitioners.

Though the early literature on externality was mainly concerned with market failure in the area of environmental pollution, the pervasive nature of market failure and its impacts are increasingly being felt in other areas of natural resource management (Panayotou, 1993) such as forests, soil and water. For instance, more developing countries nowadays adopt the 'polluter-pays-principle' (PPP) to control water and air pollution (see Mehta *et al*, 1997), the 'user-pays-principle' (UPP) in the areas of irrigation management and in other basic infrastructural projects such as water supply, sanitation and healthcare facilities. However, one of the major unresolved issues confronted by planners and policy-makers in addressing market failure is to determine the 'optimal corrective taxes' (Greenwald and Stiglitz, 1986) or fee that maximises social benefits and minimises social costs. Since the resources allocated for attaining environmental goals are generally scarce in developing countries, any policy decision to deal with market failure depends mainly on the social cost-benefit ratio which, in turn, depends heavily on the non-market valuation.

The theory of market failure was also strengthened by theoretical developments in the area of environmental benefits. More precisely, the scope of environmental benefits (in environmental economics literature) expanded over a period, which again intensified the debate on market failure (see Freeman III, 1993). Environmental benefits are classified under two major categories: use values and non-use values (Freeman III, 1993; Pearce, 1993; Pearce and Turner, 1990; Randall and Stoll, 1983). Use values are defined as values derived by individuals from the actual use of the environment (Bateman and Turner, 1993) or immediate

use of the environment (Murty and Menkhaus, 1998) or the *in-situ* use of the environment (Freeman III, 1993). The use values are further classified under two categories, namely, direct use values and indirect use values (Bateman and Turner, 1993). For instance, the direct use value derived by an individual from surface water quality may be in the form of water withdrawn for consumption whereas the indirect use value may take the form of enhanced fishing (see Mitchell and Carson, 1989). It should be noted that in developing countries, a considerable amount of use values generated by environmental resources is 'non-market use values' in nature. As already argued, a large number of households, particularly the poorer section, depends heavily on use values generated by local environmental public goods such as land, water, forests and biodiversity because they lack other kinds of economic opportunities. It should be noted that, for example, around 370 million poor people in developing countries depend on the 'marginal' agricultural areas (Barbier, 1999) and, therefore, any deterioration or overuse of this kind of 'ecologically fragile' resources has a direct impact on the very survival of these people. Rapid urbanisation, industrialisation, intensive agricultural activities, etc., in developing countries result in increased negative externality reflected in terms of point and non-point source air and water pollution, solid waste problems, soil erosion and seawater intrusion (see World Bank, 1992), which affect the quality of 'use values'. The net result is increased social cost in terms of higher mortality and morbidity, more time spent on collecting resources from alternative sources and higher defensive expenditures against pollution. Thus, all kinds of environmental and developmental projects aimed at not only mitigating the negative externalities affecting the quality of use values but also generating large-scale use values to the households will have to undergo the litmus test of cost-benefit analysis. In developing countries, many of the projects mentioned above are subject to clearance mainly on the basis of use values generated by these projects (see Pearce, 1993). However, this does not mean that one can ignore the whole bunch of non-use values at the project level (e.g., protection of wetlands; conservation of biodiversity) that constitute a considerable proportion of the total economic values in developing countries.

Even though the traditional cost-benefit analysis initially focused on estimating non-market use values (Randall and Stoll, 1983), invention of 'non-use values' of environmental resources has revolutionised the theory of non-market valuation (Freeman III, 1993), especially after the 1960s. Non-use values have many different forms. The scope of the purview of environmental benefits has expanded after the concept of option value was identified by Weisbrod (1964) and existence value by Krutilla (1967), which became important components of environmental benefits. Option value refers to the amount that individuals would be willing to pay for using a particular resource in future, in addition to their expected consumer surplus (Smith, 1987). Option value is equal to the premium that individuals are willing to pay to ensure the future availability

of an amenity (Randall, 1991). Alternatively, option value is defined as the difference between the expected consumer surplus (ES) from an amenity and the 'option price'[6] (OP) defined as the maximum amount the consumer, under conditions of demand and/supply uncertainty, is willing to pay for an option to have a resource available for use in a future time period and each year for which payment is made (Randall and Stoll, 1983; Brookshire *et al*, 1983).

Existence value refers to individuals' willingness to pay for the mere existence of a resource irrespective of its use, including the possible future use (Krutilla, 1967). The existence value, according to Krutilla (1967), is associated with two aspects: (i) individuals may be willing to pay for preserving the option for future use irrespective of whether they use it or not; and (ii) they may be willing to bequeath natural resources to their future generation. Hence, the existence value is supposed to arise out of the altruistic attitude of the individuals[7] (Freeman III, 1993).

In addition to option and existence values was the *quasi-option value* introduced by Arrow and Fisher (1974). The Quasi-Option Value refers to the value that individuals are willing to pay for delaying a decision until full information is available about the environmental amenity, while faced with information uncertainty (Mitchell and Carson, 1989). Another form of non-use value is *bequest value* arising out of the intergenerational altruism, which is an individual's willingness to pay for preserving an environmental resource that can be used by the future generation (Randall and Stoll, 1983). Even though the scope of environmental benefits expanded over a period of time, different kinds of values and their classifications led to confusion and controversy[8] (Markandya, 1998; Randall, 1991). To avoid any practical problem, the values are broadly classified into use and non-use values (see Randall, 1991) and adding up these two categories of values would provide us the "total economic value" concept (Randall, 1991; Randall and Stoll, 1983).

It can be argued that identification of the non-use values has magnified the intensity of the nature of market failure in the sense that including the non-use values in the social benefits, the divergence between private cost and social cost would expand because the non-use values shift the marginal benefits curve rightwards, under the Pigouvian framework. This justifies a higher level of tax on the polluting activities, equal to that part of value constituted by non-use values. The increased level of optimum taxation encompassing non-use values provides strong justification for increased level of investments in natural resource management. This important decision depends on the size of the total economic value estimated.

The role of non-use values in environmental management in developing countries should be addressed seriously before undertaking the valuation exercise. As far as developing countries are concerned, a considerable amount of non-use values is generated but estimating the

size of these values is constrained by the absence of a commonly accepted methodology (see Costanza *et al*, 1998). Another related question is to whom do these values occur - the individuals in these countries of origin or the individuals in advanced/developed countries? It should be noted that though the developing countries possess enormous environmental resources generating non-use values, the existing studies on this particular aspect suggest that these values are reflected in the preferences of the individuals in the advanced countries where the 'income elasticity of demand' for these values is high. For instance, in many of the studies using 'contingent valuation method', it is found that some of the standard variables such as income, education and awareness of the environmental values incorporated in the model positively and significantly influence the willingness-to-pay value elicited from the individuals. However, the conclusions of these studies should not be taken to mean that the individuals in developing countries do not have preference for the non-use values generated by resources in their own countries. It should be noted that many of the developing country CV studies suggest that the general socio-economic and institutional factors are considered to be constraining factors affecting the preferences of the individuals not only for non-use values but also for non-market use values that directly enter into the utility functions of these individuals. Even if we agree with the argument that income elasticity of demand for certain environmental resources is higher, this implies that the developing countries have to take immediate steps to protect the environmental resources since these resources potentially contribute to the welfare of the individuals when the economy attains a certain higher income level in future. Moreover, it has been empirically demonstrated that even in developing countries, preferences for non-use values do exist and they also play an important role in environmental protection measures such as water pollution control (see Markandya and Murty, 2000). Hence, estimating both the non-market use values and non-use values is an essential part of better environmental decision-making in these countries. But an empirical issue that arises here is how to incorporate the 'non-use' values into pricing of environmental goods and services, as well as into other environmental policies. Though in theory the non-use values are added up with the use values to obtain the total economic value, there are empirical difficulties in doing so, and therefore, further empirical research is needed in this area.

Theory of Policy Failure and Non-Market Valuation

Parallel to the theory of market failure, the theory of policy failure also emerged in the field of economics (see Le Grand, 1991), and it has been utilised by environmental economists to analyse some of the contemporary environmental issues (see Panayotou, 1992). This theory assumes that market failure as such (as well as the resulting environmental deterioration) arises from the failure of government policy (Maler and

Munasinghe, 1996) and is based on the argument that correcting for the 'environmentally failed policies' and strengthening the weak institutions (Pack, 1994) would itself be a step forward in mitigating the negative side-effects. This part of theory of failure gained momentum when the environmental quality in many of the Countries in Transition (CIT) which followed the planning approach to development was also found to experience severe damage (see Maler and Munasinghe, 1996). The deterioration of environmental quality in all types of countries that followed the planning approach or the market approach or both suggests that both market failure and government failure collectively as well as independently contribute to the existing environmental crisis in these countries. However, the underlying causes in both the cases are different and need separate treatment for understanding the policy implications, which is our main task in the following section.

The policy failure, according to Panayotou (1993), may arise from four different biases: (i) government's intervention in a well functioning market through subsidies, quotas, etc; (ii) government's inability to internalize the negative environmental side-effects of a policy that otherwise has a good developmental objective; (iii) government's intervention in addressing the negative externalities which nevertheless result in much worse outcome; and (iv) government's non-intervention in the market when it generates negative externality. These supply-side mis-specifications are basically supported by already existing weak institutions including the absence or the lack of a well-defined market in the relevant area, uncertainty regarding future demand and future supplies of key natural resources, the way in which societies make decisions of present against future consumption, and lack of information on the externalities caused by economic activities (Panayotou, 1992). In addition, policy-making in developing countries is immune to specific problems such as political dominance, rent-seeking behaviour, decisions dominated by professional input from usually a single discipline and lack of capacity building in other related disciplines, unreliable and invalid information used in the decision-making process and absence of effective environmental laws.

Apart from the above reasons, the policy failure has been the outcome of assumptions made in development policy-making in developing countries in the past. The major assumption underlying policy-making in many of these countries during the 70s was that there existed a 'trade-off' between development and environment, and that this was unavoidable but desirable at least in the short run. It should be noted that the environmental problems were considered as one of the developmental problems and the economic resources to address all these problems were obviously constrained by the low level of economic development. Therefore, the policies were directed mainly towards attaining a higher level of growth in gross domestic product (GDP) that was purported to

mitigate all types of problems including the ones arising from environmental deterioration (Haq, 1976). This kind of notion was also supported by the results of many empirical studies (e.g. Malenbaum, 1978; Williams *et al*, 1987), which identified the existence of the 'inverted-U-shaped curve' [alternatively called, Environmental Kuznet's curve (see Panayotou, 1993)], which is characterised by positive correlation between the per capita income and certain pollution parameters at the initial stage of development and a trade-off between them at the advanced stage. Moreover, in the early part of the 70s, policy-makers in the developing countries viewed 'environmental quality' as belonging to the advanced developed countries and, therefore, they treated it as a luxurious commodity. Increased emphasis on economic growth combined with the complete negligence of environmental issues in the policies in developing countries in the past have indeed resulted in natural resource degradation affecting sustainable development (see Maler and Munasinghe, 1996). Even during the 1990s when substantial information was available on the trade-off between environment and development, the new generation of policy-making consisting mainly of structural adjustment programmes during the 90s did not give due attention to environmental issues in countries which followed these programmes to address other developmental issues such as fiscal imbalance, balance-of-payments crisis, etc. The results of some of the empirical studies show that these programmes have even aggravated the environmental deterioration in many parts of the world (see Reed, 1996). Two points emerge from the above discussion: one is that it is not so much policy, as much as the way in which policies are framed and implemented that influences the environmental sector of an economy; second, policy-making depends heavily on information about the nature and direction of the environmental deterioration, which may not be adequately available owing to a greater level of uncertainty. This leaves the policy makers with the unanswered question of how to generate this information, and the responsibility of answering this question lies mainly with the valuation practitioners.

One more aspect to be noted is that the interlinkage between changes in the macroeconomic policies and the environmental impact at the micro-level is strong and well established (see Repetto *et al*, 1989) though not exactly quantifiable. These impacts can alter the micro-level production and consumption activities in the form of either defensive expenditure or damage cost. However, if micro-level activities fail to capture the impact of the polices due to imperfections and uncertainty then the volume of the economic as well as the secondary impacts that follow will not show up anywhere in the system. This is where the *macroeconomic valuation* (Pearce, 1993) of the environmental impact plays a role. Apparently, the macroeconomic valuation depends mainly on either the direct or the indirect market prices but, as we have already seen, they are weak in reflecting the true opportunity cost of the resource use. Even though adjustments in the estimated values are being made to

account for the subsidy factor, etc., by way of taking into account the international market prices[9], still the uncertainty about the true value of market prices persists. This being the case, valuation practitioners are confronted with the task of not only estimating the economic value of environmental changes but also choosing an appropriate economic valuation methodology.

Failure of Economic Theories and Sustainable Development

The failure of traditional macroeconomic thinking in taking into account environmental capital in its analysis has been attributed to policy failure and its associated environmental problems. This has been the major concern of many of the authors in recent years (see Repetto *et al*, 1989; Barbier, 1993). Macroeconomic thinking, after the classical economists, has been issue driven and, unfortunately, environmental issues have not been considered as macroeconomic issues until recently. The classical economists, indeed, treated income as originating from natural capital, human capital and financial capital (i.e., income is divided into three parts, namely, rent to land, wage to labour and interest to capital) and as Repetto *et al* (1989) point out, land is the central focus of the classical economic model since this is the only resource, as the classical school thought, that can experience scarcity. Barnett and Morse (1963) analysed the ideas of major classical economists separately and observed that Malthus and Ricardo, though differed in their views, were more concerned about the constraints posed by the availability of land on the overall economic development. Malthus basically assumed that the cultivable agricultural land was limited in supply and the continuously increasing population growth (in geometric proportion) accompanied by increased demand for agricultural land for food production (that grows in arithmetic proportion) would eventually result in 'diminishing returns per capita' (Barnett and Morse, 1963: 51). In Malthus' model, agricultural land was considered homogenous in nature and once the absolute limit was reached, the net outcome anticipated was 'economic stagnation', which in turn was expected to put the limit on the growth of population.[10] Ricardo, though concerned about the scarcity of agricultural land, did not foresee any kind of immediate economic stagnation imposed by scarcity of land. Ricardo basically assumed that diminishing returns were only a temporary phenomenon because land possessed heterogenous characteristics. Once the best quality land became scarce, the additional amount of land with the next best quality was to be brought under cultivation and the process was expected to enhance continuous agricultural production but with increased cost. In Ricardo's model, rent played a critical role in reflecting land scarcity. J.S. Mill, another influential classical economist, not only accepted the Ricardian-type scarcity prevailing in the agricultural sector but also believed that technology and change in the production process

in the agricultural sector play a dominant role in solving the problem of land scarcity. Mill was the first author to discuss the scarcity of non-renewable resources, especially the coal question.

It should be noted that the classical economists' approach was a broader one and the neoclassical revolution that took place towards the end of the 19th century deviated from this perspective. The neoclassical economists of the early part of the 20th century had to find solutions to unemployment, which was the major issue at that time. Therefore, they had to concentrate on the macroeconomic issues of consumption, investment, output, etc. These aspects were very well dealt with within the Keynesian macro theory, which basically relied on the circular flow model. The Keynesian circular flow model fundamentally failed to take into account the environmental sector. Since the Keynesian macroeconomic framework formed the foundation for modern macroeconomic thinking, the environment sector got completely neglected in macroeconomic policy making in almost all the countries till recently. This is where the failure of the (macroeconomic) theories is glaring.

When many of the developing countries adopted policies for sustainable development, especially after the Stockholm Conference in 1972 and the Rio Conference in 1992, they encountered crucial questions such as what was actually meant by sustainable development, the indicators of sustainable development, measurement of sustainable development, mechanisms by which sustainable development can be monitored, and so on. In other words, increased political awareness was created among nations about the need to account for the benefits and costs of environmental resources within the framework of the macro-level System of National Accounts (SNA). Some of these nations have adopted policy measures to incorporate environmental damage 'costs'/ benefits into estimation of Net Domestic Product (NDP) (see Dasgupta and Maler, 1991) so that sustainable income of these nations could be properly estimated and monitored. This basically derives from the economic theories of 'capital and income' developed especially by Hicks (1946). Despite several efforts to address the environmental issues, such as establishment of separate environmental ministries and pollution control boards, especially during the late eighties and nineties by many developing countries, it was found that the natural resource degradation and the pollution menace in these countries were on the rise, affecting the macro-level sustainable income. This being the case, many countries started developing methodologies with the help of academicians and international organisations such as UN bodies, to estimate as well as monitor sustainable income (see Repetto *et al* 1989) and these kinds of methodologies placed considerable demand on non-market valuation in developing countries.

In the seventies and afterwards, the impact of environmental problems has been realised at the macro-level and a section of

environmental economists suggested that immediate attention to be paid to some of the environmental problems affecting the sustainable development of the national economies (Pearce, 1993). Two early studies on the environment and economy linkage at the macro level - one by Barnett and Morse (1963) and another by Meadows et al (1976) - were highly influential in that direction, both showing the empirical evidence of environmental impact of development activities potentially affecting sustainable national income. The major conclusion of the Barnett and Morse (1963) study was that even though technology and substitution, the two pillars of neoclassical economics, played a role in solving the scarcity problems of exhaustible resources, the quality of renewable resources had been deteriorating. The study by Meadows *et al* (1976) stressed the importance of controlling the growth of the economy as well as of the population to contain the deterioration of environmental quality. At the political level, the Bruntland Commission's report, which publicised the concept of sustainable development, highlighted the strong linkage between natural/environmental resource use and development, and prescribed suggestions to achieve the goal of sustainable development (see WCED, 1985). Following this, many empirical studies in the late eighties and early nineties that looked into the linkage between environmental resource use and the sustainability of the economy at the macro-level found that many of the developing economies were not moving on the sustainable path (e.g., Brandon and Homman, 1995; Adger, 1993; Magrath and Arens, 1987; Repetto et al, 1989). From these studies, it was realised that sustainable development as a macroeconomic concept needed proper monitoring, which was possible only if the economic indicators were so modified as to reflect the direction of the environmental deterioration as well (see Ahmed *et al*, 1989). The underlying principle in this argument was that unless the environmental aspects were integrated with the economic aspects at the macro level, it was not possible to monitor sustainable development effectively (Lutz, 1993; Ahmed et al., 1989; El Serafy and Lutz, 1989). Towards this objective, Natural Resource Accounting (NRA) emerged as a macroeconomic tool with which one could monitor not only whether the development is sustainable or not but also derive important guidance on achieving the goal of sustainable development (Ahmed *et al*, 1989). The environmental accounting system, which is a synthesised version of environmental statistics, with the existing SNA framework, has been identified as a proper tool which can be effectively used to monitor the sustainable development of an economy (see Ahmed *et al.*, 1989; Bartelmus, 1992). Let us briefly discuss the theoretical underpinnings of sustainable development that would highlight the need for macroeconomic valuation.

Sustainable Income. Though there is a controversy over the definition of sustainable development (see O'Riordan, 1993; Redclift, 1993), sustainable income in economics can be defined in terms of 'Hicksian

income' (see Repetto *et al.,* 1989; El Serafy, 1989; Daly, 1989). Hicks defined the income of an individual as the maximum value a person can consume during a week and still expect to be as well off at the end of the week as he was at the beginning (Hicks, 1946). The basic underlying principle in Hicksian income is that the real consumption expenditure of an individual in two time periods should be equal (Turner, 1993). To maintain the real consumption expenditure, an individual has to *maintain* the 'capital stock' in such a way that a constant portion of income can be obtained over a period (Hicks, 1946). If the Hicksian individual income concept is extended to the macro level, it is equal to the NDP that is derived by deducting the depreciation of man-made capital from the Gross Domestic Product[11] (or GDP). The NDP is the net income of an economy that can be used for consumption during that particular year. However, in recent years the NDP is criticised as *not being a proper indicator of sustainable income* (see Hueting, 1980; Ahmed *et al,* 1989; Repetto *et al,* 1989; Lutz, 1993) since net national income measured in terms of NDP at present ultimately leads to eventual impoverishment (Daly, 1989). The reason is that in mainstream development economics, the concept 'capital' has been narrowly defined in the sense that it takes into account only man-made capital like buildings, machines, etc. (Repetto *et al.* 1989). Natural resources such as water, land, air, biodiversity, etc., which form natural capital (see Costanza and Daly, 1992) and their services and damages are neglected in the SNA merely because they do not carry any market price on them. Since NDP is arrived at by deducting only the depreciation of man-made capital but not that of natural capital (Costanza and Daly 1992; Ahmed *et al,* 1989; Repetto *et al,* 1989), the NDP even after deducting the depreciation for man-made capital cannot be fully consumed on a sustainable basis because it still contains considerable depreciation of *natural capital* (Daly, 1989). Even though this argument suggests that the depreciation to the natural capital stock should be accounted for in the national income accounting system, the alternative interpretation of this argument is that the natural capital needs to be maintained.

The concept of maintaining the natural capital stock has created two contrasting schools of thought, namely, weak sustainability school (also called Solow's sustainability school or Frontier economics) and *strong sustainability school*[12] (also called deep ecologist school) (Turner, 1993; Pearce, Markandya and Barbier, 1989). Both of them, one way or other, provide insights into the economic valuation of environmental resources. The weak sustainability school argues that there is no necessity for *maintaining* the natural capital stock *intact* (which is the major proposition of the strong sustainability school) but one has to maintain the overall capital stock of the economy (see Turner, 1993). This school argues that the capital stock in an economy consists of man-made capital, natural capital and human capital and, therefore, any reduction in income due to declining natural capital stock can be compensated by increasing the

man-made capital stock in the economy so that the overall income can be maintained. This implies that maintenance of the *economic value* of the *overall* capital stock itself ensures sustainable development (see Solow, 1974a; Dasgupta and Maler, 1991). The underlying assumption in this proposition is that the technological developments and the substitution possibilities between different forms of capital are considered as the major players in solving any scarcity of the natural capital stock in an economy (Barnett and Morse, 1963; Solow, 1974b). But the strong sustainability school does not agree with the argument that maintaining the overall capital stock guarantees sustainability (see Daly, 1989; Daly and Cobb, 1989). This is because the assumption that technology and substitution would solve the problem of natural resource scarcity has been found to be invalid at least as far as the renewable resources are concerned (as demonstrated by Barnett and Morse, 1963). Costanza and Daly (1992) provide two reasons why technology and substitution play a limited role in providing a solution for the dearth of natural capital. First, the empirical evidence shows that the present problems of deforestation, pollution, etc., are the result of technological changes only; and second, man-made capital is not a substitute for natural capital but natural capital is complementary to man-made capital in most of the cases (Costanza and Daly, 1992). Moreover, even if we accept the view that the possibility of substitution exists between some form of natural capital stock and the man-made capital stock, some of the services provided by the natural capital stock such as life-supporting system *cannot* be substituted with the man-made capital stock at all (Costanza and Daly, 1992). Another argument put forward by the strong sustainability school is that maintaining the *economic value* of the overall capital stock, as prescribed by the weak sustainability school, may not be practicable since economic valuation poses practical problems (see Turner, 1993). This difficulty makes compensation for the reduction in the natural capital stock impossible because determination of the amount of man-made capital required for compensating the natural capital warrants economic valuation of the reduction in natural capital, in terms of foregone welfare to society. Hence, the strong sustainability school argues that there exists a strong case for maintaining the natural capital stock *intact* (see Costanza and Daly, 1992) and this indirectly implies that the value of the natural capital stock is infinite.

If one agrees with the strong sustainability school's argument, there exists no need for economic valuation of environmental resources. The idea of the strong sustainability school, however, is criticised on two major grounds: (a) the possibility of substitution, not only between different forms of capital stock but also between different types of environmental resources within the natural capital stock, does take place and this process demands economic valuation of non-market environmental goods and services that contribute to substitution; and (b) maintaining the natural capital stock intact is possible only in a stationary state (see Solow, 1974b).

In a developing country where livelihood is dependent on the use of natural resources (Barbier, 1998), maintaining the natural capital stock intact may not be desirable. Moreover, the resource use and the underlying property rights issues contribute to the conservation of resources compared with a situation where the resource use is completely restricted. Despite the debate between the weak and strong sustainability schools, a notable aspect is that the existing debate on maintaining capital stock revolves around the level of natural capital stock that should be maintained (see Pearce, Barbier and Markandya, 1990). The sustainable development literature, in principle, has identified three levels of natural capital stock: (i) natural level of stock, (ii) optimum level of stock, and (iii) existing level of stock (Pearce, Barbier and Markandya, 1990; Pearce and Turner, 1990).

The natural level of stock is the one that is found in its natural form in the absence of any human intervention. The optimum level of stock is determined at the point where the net benefit derived from using the natural capital is maximum; in neoclassical economics it is the point at which the marginal benefit curve and marginal cost curve coincide. The existing level of stock is defined as the stock of natural resources that exists at a particular time period that may be either above or below the optimum level (Pearce, Barbier and Markandya, 1990).

The next question to be addressed is, which of the three levels of stock, is more attainable in the real world. Most of the developing countries do not possess the natural level of stock at present due to large-scale depletion and degradation (Pearce, Barbier and Markandya, 1990). The neoclassical normative approach prescribes the optimum level of stock that is based on the assumption that the benefits and costs are measured in terms of money. As we have already seen, estimation of benefits and costs is not a simple task and hence the possibility of maintaining the optimum level of natural capital stock is ruled out in practice. However, both the supporters and the critics agree that the existing level of stock in many developing countries is well below the optimum as well as natural level of stocks and, therefore, maintaining at least the existing level of capital stock is considered a necessary condition for sustainable income (Pearce, Barbier and Markandya, 1990; Pearce and Turner, 1990). However, maintaining the existing level of natural capital stock requires estimation of not only the level of stock, rate of regeneration and rate of harvesting in terms of physical units but also the economic value of the benefits generated and damages caused to the resource base for the purpose of measuring and monitoring the sustainable income. Towards this objective, NRA system as a macroeconomic tool[13] has emerged in environmental economics, which is another area that places importance on economic valuation especially in the context of developing countries.

There are two major approaches to natural resource accounting. The first is the physical accounting approach, which deals with estimation

of the stock as well as flow of resources in terms of physical units (Pearce, Markandya and Barbier, 1989). However, a major problem with this approach is that the issues addressed in terms of physical units may not be incorporated in the income accounting system because the latter deals mainly with monetary values. Hence, the monetary approach to NRA has been identified as an effective tool to address the sustainable development issues. The monetary approach is classified under three different categories, namely, (i) user cost method; (ii) net price method; and (iii) the marginal approach (or Peskin's approach). The user cost approach deals with the method of estimating that part of income from the revenue derived from exhaustible resources such as crude oil that can be invested in other investment areas so as to maintain sustainable income (see El Serafy, 1989). The user cost method is based on the assumption that there exists a possibility of substitution between natural capital and man-made capital and, therefore, extracting the man-made capital, especially, non-renewable resources, is economically justifiable if a certain portion of the sales proceeds is used to increase the level of man-made capital in the economy. The net price method simply deals with estimation of the difference between the cost of extraction and the market price of the resource concerned and tries to adjust that difference in the national income (see Repetto *et al,* 1989). The marginal or Peskin's approach utilises the economic valuation techniques that rely on the neoclassical marginal approach to place economic value on non-market benefits and costs, that can be ultimately incorporated/integrated with the income accounting system (see Peskin, 1989). Though different approaches have their own merits and demerits, all these approaches place importance on economic valuation.

Conclusion

In recent years, non-market valuation has become an important tool, playing a greater role at different levels of environmental decision-making. The available literature on non-market valuation suggests that economic valuation should be done not merely for its own sake, but rather in the interest of addressing certain fundamental theoretical issues that provide a broader framework in which such an exercise can be carried out. As a highly powerful tool in environmental decision-making, it is always important to make sure that non-market valuation is carried out in the right perspective and towards a specific goal. The major implication of our review is that economic valuation, if guided by the proper theoretical framework, will provide useful results that can be utilised for better environmental management decisions in developing countries. It is important to note that a particular kind of valuation exercise will not explain all the issues and, therefore, cannot be used for addressing different kinds of issues. For instance, the results from the contingent valuation method at the micro-level (or at the project level cost-benefit analysis)

cannot be used for addressing the same problem at the macro-level (or estimating the environmental benefits occurring from the same project at the macro-level) because of the 'adding-up problem' and the problem of benefit transformation (Griffin *et al,* 1995). Also, micro-economic valuation, based on individual preferences which are usually constrained by asymmetric or inadequate information about the environmental damages/benefits, may not be useful for deriving aggregate value at the macro level. Related to this is the problem of discounting. Though some of the empirical studies have proved that the individual discount rate and the social discount rate do not differ much, several studies have found that the individual discount rate is greater than the social discount rate. Moreover, though micro-level cost-benefit analysis and its components play an important role in addressing market failure, which ultimately places importance on economic valuation, in specific cases, such as problems where the precautionary principles could be adopted and problems which require strict command-and-control type measure, the cost-benefit analysis has only a limited role. These are some of the problems with micro-economic valuation, which could be addressed through more empirical studies. On the other hand, such methods as the replacement cost approach, which are used at the macro-level, always have the difficulty of addressing specific issues at the micro-level (benefits of restoring the entire forest cover in the country and that of a single forest generating additional non-use values are two different issues). In many cases, macroeconomic valuation becomes simple aggregation of the economic values at the micro-level. However, the value judgement that can play a crucial role in addressing equity issues at the micro-level may get neglected in macroeconomic valuation. Similarly, there may be other kinds of empirical issues that are site specific and country specific in nature and, therefore, more empirical studies need to be carried out so that the empirical issues related to micro and macroeconomic valuation could contribute to achieving sustainable development.

Notes

1 Hicksian compensating and equivalent variations refer to the consumer surplus measures which can be estimated either through individuals' willingness to pay or through willingness to accept compensation (for a detailed description of these two measures, see Bateman and Turner, 1993).

2 See Smith (1993) for an excellent review of non-market valuation studies, in general, and see Georgiou et al (1997) for a recent review of developing country based non-market valuation studies, in particular.

3 See, also, Mishan (1971).

4 A detailed description of various pollution control measures and their application is available in: Cropper and Oates (1992); Baumol and Oates (1988); Tietenberg (1985); Weitzman (1974).

5 This is because the marginal cost curve in this case lies below the level of tax, which suggests that the polluter will find it cheaper to control pollution.

6 The relationship between option value, expected consumer surplus and option price is exploited to understand the risk preference of the individuals (Hanley *et al*, 1997). For instance, if an individual is risk averse, she would be willing to pay in excess of the expected consumer surplus to avoid risk; if risk neutral, she may not be willing to pay any premium; and if risk taker, she would need to be given a premium not to take the risk (see Smith, 1987; Brookshire *et al*, 1983). However, the controversy regarding option value and option price in the literature is that which is the correct measure of value (whether option value or option price) that should be included in the cost-benefit analysis. For theoretical discussion, see Randall and Stoll, (1983). For empirical investigations, see Walsh *et al* (1984); Brookshire *et al* (1983); Greenley *et al* (1981).

7 However, the practical problem with the notion of altruism associated with the existence value is the problem of 'double counting' that may occur if the value elicited from each individual in the economy reflects not only her own benefit but also the benefit derived by others (Diamond and Hausman, 1994).

8 For instance, the distinction between the existence value and the bequest value is very vague and the option value consists of both use as well as non-use value component (see Randall and Stoll, 1983) and therefore, the major problem that would be encountered during estimation of these values is another form of double counting. Another related issue is whether option price or option value should be considered for benefit estimation (see Randall and Stoll, 1983).

9 For example, Repetto *et al* (1989) used the international market price of crude oil for estimating the cost of depreciation of petroleum in the Indonesian economy. However, the international market price of crude oil itself may be distorted since the crude oil prices are fixed by the OPEC (Organisation for Petroleum Exporting Countries) cartel.

10 It should, however, be noted that Barnett and Morse (1963) argue that though Malthus was more concerned about population growth than natural resource scarcity, he should be credited with 'a clear, forceful, and persuasive generalised statement of this view' (p.53).

11 Gross Domestic Product (GDP) is a macro concept that refers to the total monetary value of goods and services produced in an economy during a particular period, usually one year.

12 There are four different schools of thought on sustainable development (see Colby, 1990; Turner, 1993). Since they can be discussed under two broader categories, we restrict our discussion to two major schools of thought, namely, weak sustainability school and strong sustainability school.

13 Though the NRA and its components fall within the purview of 'positive economics', the information provided by them are used mainly for deriving 'normative' conclusions.

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