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Chapter 1

Plantation and non-plantation biodiversity values: distinctions of economic theories and market-based mechanisms to value ecosystems and utilisation within an Australian context.

Mark P McHenry[1]* Katinka X Ruthrof[2]

[1]Faculty of Minerals *and Energy, Murdoch University, Western Australia. 2Centre of Excellence for Climate Change Woodland and Forest Health, School of Biological Sciences and Biotechnology, Murdoch University, Western Australia*.

Abstract

Decisions regarding plantation development, either implicitly or explicitly, assign a value to ecosystems. Whilst implicit valuation simply ignores biodiversity values in plantation decision-making, explicit valuation introduces a representative value of biodiversity losses or gains. This work explores the functional components of biodiversity, the existing economic theory of biodiversity, and both advantages and disadvantages of various mechanisms that drive ecosystem valuation to further the development of market-based biodiversity policy and markets. This theoretical refinement enables both public and private decisionmakers to clarify the data requirements that underpin uncertainties in what values of biodiversity exist, to whom, and discuss options to develop a comprehensive market-based mechanism that internalises biodiversity values into everyday plantation investment decisions in the Australian context. This work suggests a hybridisation of existing valuation methods are a bridge towards functional biodiversity valuation in both plantation and non-plantation land use, These new 'non-commodity' markets may close the economic and market 'externality gap' between ecosystem conservation and exploitation, achieving conservation objectives at little cost with thoughtful land use planning.

Keywords: Biodiversity; markets; ecosystems; plantations; native vegetation.

Introduction

Article 2 in the United Nations Convention on Biological Diversity defines biological diversity as "the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (United Nations, 1992, p3). The single biggest cause of biodiversity loss in Australia is the removal, fragmentation, and degradation of native vegetation (Lockwood et al., 2000). Such land-use change is also the single biggest driver of dryland salinity and rate among the largest sources of domestic greenhouse gas emissions in Australia (Lockwood et al., 2000;

* Corresponding Author address Murdoch University, 90 South St Murdoch, Western Australia, 6150.
Email: mpmchenry@gmail.com

John et al., 2005). Nonetheless, how best to redress biodiversity loss has become much more complex than simply halting or slowing the rate of vegetative removal. While explicit economic ecosystem biodiversity valuation is uncertain and difficult, we choose to value it implicitly every day (Costanza et al., 1997), and regrettably, the implicit value is often very close to zero. Compounding the complexity of biodiversity valuation and decision-making is the lack of distinction between the public and private good, and a rigorous scientific understanding of the consequences of removal, protection, or planting of an ecosystem over time.

Economic valuation is purely anthropocentric in nature as it only considers benefits and costs relevant to human well-being. Humans benefit from natural ecosystems culturally, aesthetically, agriculturally, pharmaceutically etc., and also via the provision of such diverse services as climate regulation, soil formation, nutrient cycling, materials, fuels, quality water, etc. (Fromm, 2000). Ecosystems at the landscape level such as natural or plantation forests can yield substantial flows of economic goods and services, both before and after conversion/harvest (Balmford et al., 2002), and displacing native ecosystems with non-native ecosystems and species (e.g. the introduction of beef, wool, wheat, plantation timber production systems) can bring significant benefits to the community as whole (Bennett, 1999). These benefits ensure that there remains considerable social demand for landholders to clear or modify native vegetation for agricultural, housing, fuel, timber, plantation, or infrastructure development (Gibbons et al., 2009). On the other hand, such ecosystem modifications also result in some undesirable opportunity costs to private individuals and the general public (Bennett, 1999). At present these opportunity costs are currently ignored or undervalued in policy circles because their values are largely external to private operators (Costanza et al., 1997). Furthermore, a detrimental change in the net flow of benefits from the ecosystem (whether natural or created) eventually occurs when productive ecosystems are not managed sustainably, or the functional components are removed (Pagiola et al., 2004).

A quantification of the difference between the economic and market values of the net ecosystem goods and service flows can provide a practical means to develop a mechanism that enables outcomes that can maximise the net benefits of the ecosystem over time (Balmford et al., 2002; McHenry, 2009a). Using economic techniques to incorporate biodiversity values into decision-making processes allows a more meaningful comparison of alternative land use options of retaining, removing, modifying, or establishing ecosystems in theory (Department of the Environment and Heritage, 2005). However, in practice these economic biodiversity valuation techniques require subjective quantification and qualification of both benefits and costs (Bennett, 1999), and monetisation is often difficult and irrelevant to a market-based system at the level of the private decision-maker. However, not all net ecosystem good and service flows are external to market-based decision-making, as land values are a function of both productivity and visual amenity variables (Bastian et al., 2002). The fundamental aim of this work is to refine biodiversity value theory to increase the number of the currently few examples of plantation ecosystem values that are internalised into the market price of land (for example dryland salinity, water resources, presence of nitrogen fixing organisms in soils, etc.). These examples in many regions (not all) are insufficient to facilitate first-class land use and management priorities, and new mechanisms to internalise the various biodiversity values will be required to underpin fundamental land use change from the 'bottom up' in a market economy.

Links Between Biodiversity and Ecosystems Value

The economic value of each ecosystem arises from the interdependent relationships between species, habitat components, and their organisation that contribute to ecological functions, and the human welfare that can be derived from them over time (Fromm, 2000; Balmford et al., 2002). Thus the fundamental primary value of the dynamic evolutionary processes of the ecosystem's biodiversity, and the capability of the system to maintain stability gives rise to the secondary value of exported ecosystem goods and services (Fromm, 2000). For example, soil biodiversity research by Griffiths et al. (2000) explored the relationship between soil microbe biodiversity and agricultural pasture ecological function stability. The research found that while biodiversity does not confer ecological function stability directly, it does result in improved resilience and recovery from disturbance, and thus, continuous provision of exported services to maintain the pasture system over time. Whilst ecosystem secondary values rely on the primary ecosystem value, the primary value of one ecosystem also relies on the exported secondary values of other ecosystems (Fromm, 2000; European Communities, 2008). Therefore, the economic value to humans of exported services, or even the consumption of the primary ecosystem itself, can theoretically be traced back to the ecological biodiversity structures and functions, which in turn is derived from the ecological role of species as carriers of ecological functions (Fromm, 2000).

'Ecosystem biodiversity' refers to the variety of communities of organisms within particular habitats, and also the physical conditions under which they live, while 'functional biodiversity' refers to the existence of some redundancy in functional populations which underpin the capacity of ecosystems to absorb some disturbance without changing to a new equilibrium (Griffiths et al., 2000; Nunes & van den Bergh, 2001). As an example, the aforementioned pasture soil experimental results suggested that while the number of functional groups were important, the *level* of soil animal species biodiversity did not impact primary productivity as much as the *value* of species-specific, process-specific, and system-specific behaviour of functional groups (Bengtsson et al., 1997; Griffiths et al., 2000). Therefore, from a human welfare perspective, economic valuations should ideally focus on the relative changes in the *value* of ecosystem benefits from land use changes rather than the *level* of change in benefits flowing from the ecosystem (Nunes & van den Bergh, 2001). This is because the change in value of exported ecosystem goods and services has more of a direct impact on human welfare than changes in the level of goods and services.

As the ecological function gives rise to the ability of ecosystems to generate and export services, such as groundwater recharge, water nutrient removal, to generate economic value in theory only a limited number of physical and biological processes are required which vary in importance in different environmental conditions (Fromm, 2000; Nunes & van den Bergh, 2001). This line of logic at the first instance may seem to imply that plantation biodiversity is not a fundamental element for ecosystems in terms of economic or market value. This leads to the possibility of seriously considering the substitutability of species and their functional production values, although at present our level of knowledge of ecological interdependencies and species substitutability is far from perfect (Fromm, 2000; Farber et al., 2002). However, whilst the level of ecosystem biodiversity does not necessarily confer ecological function, it does result in a higher resilience, that is, an ability to recover quickly from disturbances (Tobor-Kaplon et al., 2005; Brussaard et al., 2007). This ability is the essence of the primary ecosystem value derived from biodiversity that has no direct economic or market value, yet is still valuable as a indirect contributor to maintaining the resilience of ecological functions and the provision of ecosystem goods and services over time (Bengtsson et al., 1997; Fromm, 2000).

Clearly, plantation biodiversity and ecological structures and functions will largely continue to be external to market economic decision-making if their value remains unquantified in a monetary sense, and clarifying the difference between total economic value of ecosystem biodiversity and the apparent market value to both private and policy decision-makers is essential. Unfortunately at present, even in theory, it is almost meaningless to ask: "What is the value of ecological support systems in total?", as their value to humans is theoretically infinite (Costanza et al., 1997). However, it is economically meaningful to ask "How value changes in the quantity or quality of ecosystem services may impact human welfare directly and indirectly?", to be able to represent a total economic value (Costanza et al., 1997; Farber et al., 2002). Nonetheless, this work argues that it is more practical in a market economy to also ask: "How value changes in the quantity and quality of ecosystem services from a particular area of land impacts the welfare of individual decision-makers in financial terms?" Attempting to answer this question provides a real market value, sidestepping the uncertainties of economic values, and/or enabling comparisons between the plantation ecosystem's economic and market value.

Indirect and Direct Biodiversity Value Theory

Clarification of existing theory enables a clearer theoretical basis to integrate both positive and negative externalities of activities that influence the level of change in benefits flowing from forestry ecosystems. The identification of total economic value generated by natural assets recognises the anthropocentric, instrumental, and utilitarian values that are gained or lost by segments of the environment that affect the welfare of at least one private individual directly. This includes the biodiversity value gained or lost by segments of the environment that affect the prices of agricultural and forestry inputs and products, in addition to the productive use of species and genetics in these industries. Separate from direct productive value, ecosystem biodiversity also has direct value for individuals in terms of aesthetics, recreation, or simply for it to exist, whether or not it is utilised now, in the future, or at all. However, the inclusion of ecological structures and function is necessary for total economic value assessments, as individual and production values of biodiversity do not recognise often indirect and complimentary relationships between humans and ecosystems (Fromm, 2000).

Indirect use values of biodiversity are associated with ecosystem infrastructure that supports economic activity (Nunes & van den Bergh, 2001). Indirect uses include functional benefits for life-support ecological functions through the provision of soil formation, climatic stability, clean air and water etc. Indirect ecological functions such as regulation of climatic processes, the hydrological cycle, processing of human induced pollution, (etc.) can be viewed as avoided health and material possession damages, and can be calculated as indirect benefits which can be quantified probabilistically using aggregated data akin to a form of insurance. The value in this case arises from the ecosystem protecting human capital, human-made capital, and natural capital against disturbances. These indirect values of ecological structure and function transcend the simple value of inputs for production (such as plantation timber) and the value of an individual ecosystem itself (Fromm, 2000). Some indirect use values manifest themselves as direct use values (Gilespie, 2000), especially in plantation and agricultural production systems, such as improved stock water quality and storm protection. Indirect use values can even include biological resources used to produce goods and services such as pharmaceuticals (Nunes & van den Bergh, 2001). In contrast, direct use values of biodiversity often refer to human uses of biodiversity in production and consumption, which can also include tourism, research, and other activities (Gilespie, 2000; Nunes & van den Bergh, 2001). A market analogy for direct and indirect use values are human-made assets for direct individual use (i.e. timber, furniture, houses) and productive assets (i.e. tools, plantations, farms), which are protected by security assets that support activities indirectly (i.e. private liability insurance and social welfare systems) (Fromm, 2000).

The reductionist approach of determining total economic value disaggregates biodiversity into more categories to calculate the total economic value as the sum of various use and non-use values with a bottom up approach (Nunes & van den Bergh, 2001; Hecht, 2005). See Equations 1 and 2. Indirect use values include vicarious values which relate to the benefits of indirect consumption through books, documentaries, and other media. Non-use values relate to benefits individuals obtain from the resource without directly or indirectly using them, and include existence values, option values, quasi-option values, and bequest values (Gilespie, 2000). Existence value is simply the benefits from knowing that certain things remain conserved and certain species and ecosystems survive (Bennett, 1999; Gilespie, 2000). Option values relate to the maintenance of the right to use a resource, without necessarily doing so, while quasi-option values refer to the benefits obtained from the opportunity to delay decisions to make the most of improved information about the resource over time. Finally, bequest values refer to the maintenance of environmental attributes for future generations (Gilespie, 2000). These non-use values have the capacity to reflect human, moral, philanthropic, or policy considerations of biodiversity protection intergenerationally (Nunes & van den Bergh, 2001). Whilst these values are all valid, they pose significant difficulty to market-based mechanisms and policymakers who may rely on balancing both development and conservation pressures.

Equation 1. Total economic value. Source: (Hecht, 2005).

Total economic value = use value + non-use value

Where, use value = direct use value + indirect use value, and; non-use value = existence value + option value + quasi-option value + bequest value.

Equation 2. Total economic value simplified. Source: (Hecht, 2005).

Total economic value = direct use value + indirect use value + existence value + option value + quasi-option value + bequest value

Formal valuations of ecosystem goods and services provide insights into decision-making trade-offs for or against ecosystem conservation, modification, or establishment (Howarth & Farber, 2002). The aim of valuations are to clarify the value of the trade-offs between the productive benefits and environmental benefits when land is utilised for, or taken out of production (Bennett, 1999). The assessment of trade-offs require specialist knowledge of the ecosystem, and the economic decision-making regarding conservation and production to consider all potential benefits and costs generated by the natural resource (Fromm, 2000). A simple and practical example is from the perspective of administration and monitoring, where the benefits of environmental monitoring to underpin biodiversity indicators and markets should exceed the costs by the greatest absolute amount (Pannell & Glenn, 2000). Yet, an expansion of quantifying benefits and costs in terms of the economic value of the actual biodiversity goods and services are more complex, as it attempts to apply worth to ecological structure and function (Fromm, 2000). This is required to give policy-makers options, and also a rationale to protect non-plantation forestry assets, as conservation planning never occurs in isolation from politics and economics (Fromm, 2000; Polasky et al., 2005).

The Monetary Value of Biodiversity

Economic valuation of biodiversity strives to overcome the current economic causes of biodiversity loss and to ensure economic incentives are established to encourage biodiversity conservation (Emerton, 2001). Economists consider that particular choices are desirable if the benefits to the community exceed the costs from a community perspective (Gilespie, 2000). Ecological economists are involved specifically with the relationships between property rights and resource management, and model the interactions between the economy and the environment, and use new instruments of environmental policy (Martinez-Alier, 1990). These new instruments have been developed to correct many existing market failures that do not account for the costs of biodiversity or ecosystem loss (Pagiola et al., 2004). These land use planning market failures drive biodiversity and habitat loss by discounting or excluding non-market benefits in market-based plantation decision-making (Balmford et al., 2002).

Market mechanisms such as carbon prices, biodiversity credits, or premium pricing for sustainably produced goods and services, capture ecosystem values at a private level for producers to allow them to have an incentive to generate positive outcomes (Balmford et al., 2002; McHenry, 2009a, 2009b, 2010, 2011b, 2011a, 2012a). When these market-based measures have been developed and implemented well, they enable sustainably-produced goods and services to compete with conventional products that are effectively subsidised through depreciating natural ecosystems (McHenry, 2011a, 2011c). Many ecosystem services do not qualify for market trading as they are not private in nature (Farber et al., 2002). Conserving relatively intact habitats on private land alongside timber plantation and other production systems will often require compensatory mechanisms to mitigate the negative private impact (Balmford et al., 2002). This is because some of the economic value of native vegetation accrue to the broader community, while the associated costs of maintenance fall on the landholder (Gilespie, 2000). Even when compensatory mechanisms do exist, smaller incentives to landholders may be regarded as a waste of time, or even a direct insult to the private owner (Lockwood et al., 2000; McHenry, 2012b), and the high level of work involved in actively maintaining conservation areas is often underestimated by the broader community, including decision-makers.

Options amenable to landholders, such as tax reductions, or exempt status, low-interest loans, grant schemes, and other associated financial mechanisms could assist non-plantation vegetation management on private lands (Gilespie, 2000). In reality, there are a number of reasons why landholders remove, degrade, retain, improve or plant vegetation, which may or may not be related to financial benefits, or optimal for the wider society (Gilespie, 2000). Therefore, the use of both economic and informative mechanisms may have an improved chance of assisting landholders to compare their available options, while including the real value of the vegetation (both non-plantation and plantation) to the society.

The existence of land use externalities, (for example submerged ecosystems and altered river flow regimes from the construction of very large water supply dams) form of market failure, and a committed government can minimise their distortionary impact on the community and the environment

(Gregory Mankiw et al., 2000; Foxon et al., 2005; Jaffe et al., 2005; McHenry, 2009a). While private businesses understandably do not invest in goods and services such as clean air and water that are often free, market mechanisms can reimburse entities for protecting the quality of goods and services while at the same time regulating unacceptable outcomes (Longo & Markandya, 2005). This has developed in the energy efficiency market with the introduction of regulatory minimum performance standards to exclude inferior appliances, while information instruments allow consumers choose to pay a premium to obtain products exhibiting high energy and water use efficiencies. Therefore, a neat distinction cannot be made between market and regulatory measures, as all market-based measures require a regulatory and institutional setting (Diesendorf, 2007).

Establishing formally protected lands through regulation-only mechanisms may conserve habitat, but socio-economic and political constraints limit this form of ecosystem conservation (Polasky et al., 2005). The economic foundation of a decision for, or against, the protection of biodiversity requires the inclusion of all costs and benefits relating to it. However, there is an 'externality gap' between the market and economic value of biodiversity. Filling this valuation gap requires the identification of, and where possible, the monetisation of the services that vegetative asset provides (Fromm, 2000). Economically biodiversity must be seen as an asset, and biodiversity conservation as an investment (Fromm, 2000; Farber et al., 2002). Neglecting conservation can be interpreted as de-investment in assets, which in turn leads to a reduction in ecosystem service provision, which leads to an economic cost (Fromm, 2000). Ensuring the continued provision of ecosystem services requires conservation of natural systems, which also in turn calls for economic valuation (Balmford et al., 2002).

Using monetary indicators for economic valuation of biodiversity enables comparisons of alternative market-based ecosystem management options, while non-economic assessments of biodiversity values do not (Nunes & van den Bergh, 2001; Pagiola et al., 2004). Monetary indicators offer the flexibility to be based on various market price valuations or even individuals willingness to pay for such services. However, with this flexibility comes the potential to derive ambiguous monetary values of biodiversity, as different valuation methodologies each have their strengths and weaknesses (Nunes & van den Bergh, 2001). While a reasonable level of flexibility is necessary for assessment methods to accommodate unusual situations, this flexibility needs to be balanced to ensure environmental considerations are not compromised (Gibbons et al., 2009). At times, ecosystem, economic, and market values are at odds with each other, as only some of the species in an ecosystem are valued due to a number of reasons (Farber et al., 2002). It is for this reason that economic biodiversity indicators and methods ought to be based on accurate biological indicators based on scientific principles (Nunes & van den Bergh, 2001).

Market and Non-Market Biodiversity Valuation

The main obstacles to the wider application of biodiversity valuation in Australia are lack of biophysical information to support valuations, the technical accuracy of valuation techniques, and ethical concerns over valuing environmental impacts in monetary terms (Department of the Environment and Heritage, 2005). Particular ethical criticisms of economic biodiversity valuations relate to conferring of dollar figures on 'priceless' biodiverse assets, such as a non-plantation forest or river ecosystem. As always, there is a counter argument: human development decisions either implicitly or explicitly value ecosystems. Implicit valuation simply ignores biodiversity values in decision-making, and, by comparison, explicit economic valuation represents the potential biodiversity losses (Department of the Environment and Heritage, 2005). The primary purpose of economic valuation is to obtain consistent information on the costs and benefits of biodiversity conservation to inform decision-makers (Pagiola et al., 2004). This may balance the predominance of implicit valuation processes.

There are a variety of values that biodiversity can be attributed beyond individual and productive values, including security values that ensure the continued service provision from ecological functions (Fromm, 2000). Theoretically, biodiversity value can be characterised by a number of values: local versus global diversity, life diversity versus biological resources, instrumental versus intrinsic values, and so on (Nunes & van den Bergh, 2001). Biodiversity value can also be categorised in terms of an ecosystem spatially, or a habitat that is in high demand, such as areas of recreation or tourism (Nunes & van den Bergh, 2001). However, policymakers must be aware that there are unresolved issues in some valuation methods. For example, whether economic valuations at multiple levels leads to double counting of biodiversity values (McHenry, 2011a).

Market-based classifications of economic valuation include techniques such as: the human capital approach; productivity changes method; defensive expenditures; repair/replacement expenditures; shadow projects, and; the opportunity cost method (Gilespie, 2000). In comparison, there are six major ecosystem service non-market economic valuation techniques when market valuations do not capture the social value of biodiversity: avoided cost techniques quantify the value of costs that would have occurred in the absence of certain ecosystem services; factor income techniques value the enhancement to incomes from improving ecosystem services; travel cost techniques reflect the costs people are prepared to pay to travel to enjoy ecosystem services of specific regions; hedonic pricing techniques reflect the differential prices people pay for goods that involve specific ecosystem amenities; contingent valuation techniques value ecosystem services by quantifying the differential values that people are willing to pay for hypothetical ecosystem service alternatives, and finally; replacement cost techniques use the cost of substitutes that can replace the ecosystem services (Farber et al., 2002).

The replacement cost technique is the only technique in both-market and non-market categories, as it leaves scope to sum additional techniques to derive site-specific non-market values (such as travel costs) and likely market-based values (such as plantation carbon sequestration). The replacement cost technique estimates how much it would cost to replace an environmental resource and is a promising approach to provide a substitute for an ecosystem service valuation technique (Department of the Environment and Heritage, 2005). The replacement cost technique gives value to ecosystem services by quantifying the cost of restoring or synthetically replacing it (Balmford et al., 2002). This technique does not strictly evaluate the value of biodiversity benefits, but is useful for providing an initial estimate of the resources value (Department of the Environment and Heritage, 2005). While intuitively appealing, replacement cost methods may misrepresent the 'willingness to pay' or 'willingness to accept' valuation concepts in some circumstances where social amenities are lost in the synthetic replacement (Farber et al., 2002). Nonetheless, replacement cost methods do provide an easily verifiable and practical methodological choice to reveal the lower bounds of biodiversity value to restore functional diversity and facilitate a cost-benefit analysis of land use change options.

Formal cost-benefit analyses of areas of biodiversity can be used to determine productive and individual values, although essential services may not be considered when there are significant ecosystem knowledge gaps (Fromm, 2000). For example, a robust attempt at valuing flood protection services provided by various vegetative islands (both plantation and non-plantation) must be based on complex hydrological models of topography and ecosystems, and uncertainties or errors can become considerable (Howarth & Farber, 2002). However, if the economic costs of establishing and maintaining vegetative islands on degraded land is low, then there is little practical barrier to 'over-engineering' to ensure sufficient protection as a form of insurance. If a plantation project aimed to replicate the original high-quality habitat, then this organic category of replacement cost method would have roughly comparable ecological function in terms of exported ecosystem services as the original habitat (Emerton, 2001). Replacing the original vegetation would also avoid the lost social benefits and may more accurately represent 'willingness to pay' or 'willingness to accept' values by improving, retaining, or re-establishing ecosystem services over and above flood-protection (Farber et al., 2002). However, economic valuations such as replacement cost tend to handle large-scale and long-term problems poorly, but have the potential to be suitable for looking into shorter-term and local-scale values (Pagiola et al., 2004).

Ecosystem Service Valuation Method and Limitations

Valuation studies illuminate ecological structure and function relationships and their roles in supporting human welfare (Howarth & Farber, 2002). While economic valuation has both strengths and limitations as a decision-making tool, it is clear that information about environmental management costs and benefits are essential to ensure efficient, equitable, and sustainable outcomes. While most of the direct and indirect use values of ecosystems may be approximated quite accurately, the availability of physical data or the change in the functional ecosystem services are often limited (Pagiola et al., 2004). Variables, such as vegetation condition, percentage of vegetation types cleared in the region, and the area of any potential vegetative offset location should receive special attention during assessment (Gibbons et al., 2009).

Local historical and cultural knowledge of ecosystems and their traditional land uses is also recommended to inform biodiversity valuation studies (Dyer et al., 2008). When cultural, historical, and social systems are intimately entwined with ecosystems, the individual component values should, in theory, be a larger value than the sum, as these values are more communal and have greater interpersonal impacts than standard economic ecosystem values (Farber et al., 2002). Therefore, there is no one 'correct' method or technique to obtain ecosystem values, and there is a need for, as Farber et al. (2002) p390 describes as "conceptual pluralism, and thinking outside the box" in its development. These issues beg the question of: "How valuable are ecosystems to whom?", as ecosystem benefits can fall unequally across different groups of people, while being valuable to some and incurring costs to others (Pagiola et al., 2004).

Landscape, species, and genetic diversity that provide input into productive processes have been widely valued using the contingent valuation method (Nunes & van den Bergh, 2001). Contingent valuation asks how much a person would pay for a particular environmental outcome, or how much compensation they would be prepared for its loss (Balmford et al., 2002). The contingent valuation method is the most useful to identify and measure economic non-use values. In principal, the contingent valuation method is applicable for all biodiversity categories, except for categories that the general public is not informed about, or has little experience with, for example: ecosystem life-support function valuations. When far removed from human perception, contingent valuation becomes problematic when eliciting the economic value of ecological processes (Nunes & van den Bergh, 2001), such as the carbon fixation of trees, or respiration of soil biota. Contingent valuation is more suited to interpretations of existence and bequest values from the amount an individual would pay to know that a particular native fish exists in its natural habitat and remain so for future generations, respectively. The concern with contingent valuation is the reliability and validity of the responses (Loomis et al., 2000). These issues may be improved by including the hedonic price method, where environmental services are valued by comparing market prices of biodiversity conservation at a regional scale, such as a water body or catchment (Lockwood et al., 2000; Balmford et al., 2002).

A hybridised total economic valuation of use and non-use values could be used by utilising government departments expenditure on specific ecosystem amenities (the hedonic method) and the additional costs that people are prepared to pay to travel to use the ecosystem (travel cost method), with the addition of a contingent valuation study of the non-use values of native vegetation that sum to add value to the construction of new habitats (replacement cost method). Using hybrid methods enables flexibility to cater for the unique circumstances of each biological system (both plantation and non-plantation), and introduces a higher level of rigor for decision-makers when choosing between various direct, indirect, or non-use alternatives, than simply the current status quo of implicit valuation.

Conclusion

Regardless of the level of scientific rigor, high-precision, or accuracy of data utilised to underpin and verify biodiversity values, there will necessarily remain a subjective human and local element to the economic values determined. Rather than a stark trade-off between biodiversity conservation and high-value plantation commodity production, a large fraction of conservation objectives can be achieved at little economic cost with thoughtful land-use planning (Polasky et al., 2005). While economists continue to debate the validity of economic valuation methods, rightly or wrongly, they undermine the public confidence in valuation techniques (ten Kate et al., 2004). At the same time, questions of irreversibility and uncertainty raises issues for environmental valuation (Howarth & Farber, 2002). The prime reasons for the explicit valuation of biodiversity is to introduce at least some value into decision-making, and simultaneously foster a level of rigor in the analysis of the costs and benefits of various alternative options available.

This review suggests that valuations of functional biodiversity is a bridge towards market valuation of biodiversity, and new 'non-commodity' markets may close the economic and market 'externality gap' between ecosystem conservation and exploitation. However, functional biodiversity is difficult to value. Thus, the development of a hybridised 'total economic valuation' approach considering both use and non-use values of government expenditures on specific ecosystem amenities and the travel costs that visitors incur, alongside a contingent valuation of the non-use values of vegetation, and replacement cost methods for new plantations, may a be a suitable approach. Such hybrid methods enables flexibility to cater for each biological system (both plantation and non-plantation), and introduces a level of relative comparison for decision-makers that is at least an improvement on the current status quo of implicit valuation.

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