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Clem Tisdell
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Clem Tisdell[2]

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Dayuan Xue[3]

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[2] School of Economics, The University of Queensland, St. Lucia Campus, Brisbane QLD 4072, Australia
Email: c.tisdell@economics.uq.edu.au

[3] School of Environmental and Life Science, Minzu University of China, Beijing, China.
Email: xuedayuan@hotmail.com

The *Economics, Environment and Ecology* set of working papers addresses issues involving environmental and ecological economics. It was preceded by a similar set of papers on *Biodiversity Conservation* and for a time, there was also a parallel series on *Animal Health Economics*, both of which were related to projects funded by ACIAR, the Australian Centre for International Agricultural Research. Working papers in *Economics, Environment and Ecology* are produced in the School of Economics at The University of Queensland and since 2011, have become associated with the Risk and Sustainable Management Group in this school.

Production of the *Economics Ecology and Environment* series and two additional sets were initiated by Professor Clem Tisdell. The other two sets are *Economic Theory, Applications and Issues* and *Social Economics, Policy and Development*. A full list of all papers in each set can be accessed at the following website: http://www.uq.edu.au/economics/PDF/staff/Clem_Tisdell_WorkingPapers.pdf

For further information about the above, contact Clem Tisdell, Email: c.tisdell@economics.uq.edu.au

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Managing Ecosystem Services for Human Benefit: Economic and Environmental Policy Challenges

ABSTRACT

In the Millennium Ecosystem Assessment (2005), ecologists identified and pointed out a multitude of environmental and other benefits obtained by human-beings from ecosystem services. Frequently, these benefits are not fully recognized and they are not adequately taken into account in decision-making in contemporary economic and political systems for reasons outlined in this contribution. In particular, this adversely affects the optimal conservation of natural, near natural and unmanaged ecosystems. The human benefits from ecosystem services as set out in the Millennium Ecosystem Assessment are summarized and this assessment is examined critically. Economic views about the economic value of different types of ecosystems and forms of biosphere use are outlined and assessed.

Determining the economic value of alternative forms of land-use (more generally biosphere-use) is extremely difficult because of knowledge constraints. Often the biophysical consequences, that is, variations in the supply of ecosystems services resulting from alterations in ecosystems, are poorly known. The economic valuation of changes in ecosystems (alterations in biosphere-use) is also hampered by poor information about the demand for these services (for example, the willingness of beneficiaries to pay for their supply) and the cost of replacing these services if they are lost (or diminished in availability) as a result of ecosystem change. While this limits the scope for economic valuation, it does not mean the rational valuation of biosphere use is impossible. It has been suggested that the supply of ecosystem services can be managed optimally, in some cases, if private landholders are paid for supplying these services. The benefits and drawbacks of this approach are discussed. China’s policies to restore the supply of particular ecosystem services, for example, its Grain-for-Green program, are used to illustrate some of these matters.

JEL Classification: Q57

Managing Ecosystem Services for Human Benefit: Economic and Environmental Policy Challenges

1. Introduction

The many services provided by ecosystems contribute significantly to human well-being and play a vital role in conserving biodiversity (Millennium Ecosystem Assessment, 2005). Despite this, many of these services are being lost, diminished or degraded, thereby reducing human welfare, because of deficiencies in contemporary economic and political systems. These deficiencies seem to be reinforced by a lack of awareness among policy-makers and members of the general public about the value of these services. Furthermore, human alternatives to some ecosystems can potentially result in unsustainable economic development, and diminish the incomes of future generations.

While determining the use and conservation of ecosystems so that they provide maximum human benefits is an important task it is also challenging for reasons outlined in this chapter. This is not only because the measurement of human welfare depends on varied moral or value judgments. Even if an appropriate measure of human welfare is agreed upon, there are difficulties in obtaining adequate biophysical data and utilizing it to obtain the optimal solutions to ecosystem use and conservation. There are bounded rationality problems (Simon, 1957; Tisdell, 1996, Ch. 1) and even optimally imperfect decision-making of the type described by Baumol and Quandt (1964) can be difficult or impossible. Despite this, scope does exist for improving decisions about the use and conservation of ecosystems or more broadly, human uses of the biosphere.

In order to do this, several pertinent matters need to be considered. These include the identification of ecosystems services of value to human beings. This is given considerable attention in the Millennium Economic Assessment (2005). This assessment is discussed critically in this chapter. Economists have also assessed the value of ecosystem services both on global and local scales and selected economic articles on this subject are reviewed. This is followed by an outline and discussion of reasons why economic and political systems display failures in conserving and managing the supply of ecosystem services. However, it is very difficult to determine the economic value of ecosystems and changes to these. Reasons for this are given and discussed. It has been suggested that the supply of ecosystem services can be improved if landholders are paid for their provision. The benefits are drawbacks of this approach and considered. This is followed by an outline and discussion of some of China's policies for restoring ecosystem services.

2. The Millennium Ecosystem Assessment and Its Limitations

The Millennium Ecosystem Assessment (2005) was a major task undertaken by ecologists. According to Norgaard (2010, p. 1220) over 1400 scientists from around the world contributed to this assessment over a five-year period. They identified numerous ecosystem services of value to human-beings, the state of these services, and factors altering ecosystems. Services provided by unmanaged ecosystems (for example, natural and near natural ecosystems) as well as by managed ecosystems (for instance, agricultural systems) were identified. These services were classified into four categories:

1. Provisioning services
2. Regulating services
3. Cultural services
4. Supporting services

Provisioning services cover mainly material benefits to human-beings (for example, provision of food and fiber), regulating services encompass environmental services that are of value to humans, cultural services include spiritual and religious values and supporting services include such services as nutrient and water recycling. This list of services (together with examples) is presented in Table 1.

Table 1. Types of services provided by ecosystems according to the Millennium Ecosystem Assessment (2005)

Category of Service	Examples
1. Provisioning services	Food/fiber; genetic resources; biochemical, natural medicines and pharmaceuticals; ornamental resources; fresh water.
2. Regulating services	Air quality regulation; climate regulation; water regulation; erosion regulation; water purification and waste treatment; disease regulation; pollination; natural hazard regulation.
3. Cultural services	Cultural diversity; spiritual and religious values; knowledge systems; educational values; inspiration; aesthetic values; social relations; sense of place; cultural heritage values; recreation and ecotourism.
4. Supporting services	Soil formation; photosynthesis; primary production; nutrient cycling; water cycling.

Source: Based on Millennium Ecosystem Assessment (2005, Table 2.2, pp. 33-37).

While this classification brings attention to ecosystem services that are likely to be valued, the assessment does not give much attention to their comparative value. Furthermore, the report makes no mention of disservices to human beings which are associated with some ecosystems. For example, some ecosystems provide breeding grounds for pests. For example, mosquitos breed in swamps and transmit malaria. Protected areas often harbor wild animals which attack the crops and domesticated animals located near such areas. Such ecosystems are a source of fires which cause economic damage and economic losses are associated with damaging floods associated with unregulated river systems. Furthermore, some unmanaged ecosystems hinder transport and communication. Therefore, it needs also to be recognized that ecosystems can generate negative effects which human beings try to limit by engineering and other means. Usually, both benefits and costs to human beings are associated with particular ecosystems and changes to these. It could therefore, be claimed that the Millennium Ecosystem Assessment is not very balanced because it focuses mainly on the benefits from such systems.

The Millennium Ecosystem Assessment does not provide guidance on the economic valuation of ecosystems nor does it make concrete suggestions on how ecosystems might be optimally managed, changed or conserved. Economists have, however, produced some estimates of the economic valuation of ecosystems (see the next section). However, economic methods do not enable the economic value of most cultural services to be measured but economic methods are available to measure a few, e.g. the economic value of recreation and nature-based tourism (see, for example, Tisdell, 2005, Ch. 7; Tisdell and Wilson, 2012; Xue, *et al.*, 2000). While those framing the Millennium Ecosystem Assessment (2005) do list in Table 5.1 (pp. 78-80) desirable goals and targets for ecosystem conservation, these seem to be primarily opinion or consensus-based.

An ecosystem has been defined "as a community of living organisms (plants, animals and microbes) in conjunction with non-living components of their environment (things like air, water and mineral soil) interacting as a system" (Anon, 2013, p.1). A difficulty in determining the boundaries of ecosystems is that their boundaries are often unclear. It has been pointed out that "classifying ecosystems into ecologically homogenous units is an important step towards effective ecosystem management, but there is no single agreed way to do this" (Anon, 2013, p.1). Furthermore, human alterations to pre-existing ecosystems add to their heterogeneity and make it more difficult to establish homogenous sets of ecosystems. In addition, the functioning of some ecosystems depends to some extent on others or are interdependent (see, for example, Azqueta and Sotelesk, 2007). For instance, the functioning of most aquatic ecosystems depends on the functioning of terrestrial ecosystems. Therefore, when evaluating an ecosystem, discretion has to be exercised in determining its boundaries and the extent to which its interconnections with other ecosystems should be taken into account.

3. Estimates of the Economic Value of Different Ecosystems

Costanza *et al*. (1997) presented one of the earliest estimates of the economic value of ecosystems. They estimated the economic value of ecosystems for the world as a whole and stressed the value of natural capital which produces many of these services. Their ambitious estimation project drew on and projected the results of more restricted economic studies of the value of ecosystem systems.

Not surprisingly they found that the economic value of the annual flow of ecosystem services exceeds the total value of global GDP and is about twice as large as global GDP. This is to be expected because many of the services provided by ecosystems are not marketed and therefore, their economic value is not counted in GDP.

Seventeen types of ecosystem services were taken into account with as much information as possible being provided for the economic value for 16 biomes. The biomes were classified according to whether they are marine or terrestrial. It was estimated that economic value of annual ecosystem services provided by marine biomes is about two-thirds larger than that for terrestrial biomes.

The estimated value of each type of ecosystem service is presented for each of the 16 biomes by Costanza *et al*. (1997, p. 256) in matrix form in their Table 2. Several cells in this table are empty (open) because of insufficient information and some are shaded. It is stated that the shaded cells "indicate services that do not occur or are known to be negligible" (Costanza, *et al.*, 1997, p.256). However, some of the entries seem to be inaccurate. For example the item, genetic resources is shaded for cropland. This indicates that cropland plays no role or only a negligible role in supplying genetic resources. Despite this, many crops ensure conservation of agro-biodiversity, much of it of a heritage nature and there is considerable concern that much of this biodiversity is being irreversibly lost (Tisdell, 2012a). Therefore, this entry is inaccurate. Furthermore, the genetic resources provided by grass/rangelands are designated as being of zero value. This is hard to believe because the wild relatives of some grain crops, such as wheat, occur in grasslands. Furthermore, the comparative economic value attributed to the ecosystem services of cropland and to grassland/rangeland appear to be too low. They are attributed respectively with only 0.39% and 2.73% of the annual value of ecosystem services produced. In addition, their relative contribution to the value of food production seems, on the face of it, to be too low. Only 3.9% of the value of total food production is attributed to crop land and 4.83% to grassland/rangeland.

The results indicate that natural or near natural systems accounts for the lion's share of the economic value of ecosystem services. While this is most likely so, it also seems to be the case that the comparative value of ecosystem services provided by managed ecosystems has been under estimated. However, the main purpose of this exercise seems to have been to demonstrate that the ecosystem services generated by natural and near natural ecosystem have a high economic value, the extent of which often fails to be appreciated because most of these services are not marketed.

A much more recent estimate of the global economic value of different ecosystems and their services was completed by de Groot *et al.* (2012). Like the study by Costanza *et al*. (1997) it relied on secondary data. These data were sourced from the Ecosystem Valuation Database developed for the Economics of Ecosystems and Biodiversity (TEEB) studies. Unlike in the classification by Costanza *et al*. (1997), de Groot *et al*. (2012) classified the benefits of ecosystem services in the same way as that adopted in the Millennium Ecosystem Assessment. Monetary values were estimated for ten biomes and presented as average values per hectare per year (de Groot, *et al*., 2012, Table 2, p. 55). Thus fewer biomes were covered by de Groot *et al*. than by Costanza *et al.* (1997). Cropland was not included but grasslands were. Once again, no value was attributed to the genetic resources contained in grassland but a large value was assigned to the value of genetic diversity in grassland. On a per hectare basis, the highest annual economic value was attributed to coral reefs followed by the following in declining order: coastal wetlands, inland wetlands, tropical forest, fresh water (rivers/lakes), temperate forest, grasslands, woodlands and marine ecosystems. Once again, natural or near natural ecosystems were shown to have the highest annual monetary values per hectare.

However, studies of the type completed by Costanza *et al*. (1997) and de Groot *et al*. (2012) only have limited direct practical policy applications. They can mostly be regarded as attempts to raise awareness about the economic value of natural or near natural ecosystems. They do not, for example, indicate whether the conversion or partial conversion of some ecosystems raises or decreases economic welfare. There is virtually no concrete discussion of the opportunity costs nor economic trade-offs involved in land conversion. It is likely however, to be difficult to do this at a global level, but it can be done at a localized level. It is desirable to do this from a pragmatic point of view. De Groot *et al*. (2012) point out that their results show that most of the economic value of ecosystem service is outside the market and state: "given that many of the positive externalities of ecosystems are lost or strongly reduced after land conversion better accounting for public goods and services provided by ecosystems is crucial to improve decision making and institutions for biodiversity conservation and sustainable development". While this is true, this finding does not provide specific policy guidance. Guidance on the economics of alternative forms of land-use is likely to be easier to provide at the local or regional levels.

A more localized economic valuation of ecosystems services was completed by Xue and Tisdell

(2001). They evaluated the economic value of the ecosystem services provided by the Changbaishan Biosphere Reserve in Northeast China using mostly the cost of replacing these services as a measure of their economic value. The services taken into account included water conservation, soil protection, CO_2 fixation and O_2 release, nutrient recycling, SO_2 absorption, and disease and pest control. It was found using this method that the economic value of these services is approximately ten times the economic value of this reserve if it were to be used for logging. It was assumed that most of the ecosystem services provided by this reserve would be lost if it were to be used for commercial logging. This is, however, probably too strong an assumption, the demand for some of the ecosystem services may be less than their supply and the demand for some may be less than the cost of replacing them (Tisdell, 2012b). Therefore, these estimates may overstate the economic value of conserving this reserve. On the other hand, Changbaishan Mountain Biosphere Reserve is extensively used for ecotourism and its economic value for this purpose has not been factored into the estimates by Xue and Tisdell (2001), although Xue *et al*. (2000) estimated this value using the travel cost method. While the value of the reserve for ecotourism would be diminished by its use for commercial forestry, it still might attract some tourists depending on where forestry is practiced in the reserve.

The main purpose of this article by Xue and Tisdell (2001) was to increase awareness among Chinese policy-makers that because of the ecosystem services provided, the economic value of conserving some natural or near natural areas exceeds their economic value when converted to commercial (market-based) uses. It was, nevertheless, stressed that methods for the economic valuation of ecological functions need considerable improvement.

Since 2000, some progress has been made in assessing techniques for determining the economic value of ecosystem services (see, for example, Pagiola, *et al.*, 2004) but further advances would be useful, especially the development of techniques that take into account the bounded rationality problems involved in these complex evaluations. Ninan (forthcoming) provides a coverage of recent case studies of the economic value of ecosystem services and discusses some of the methodological issues involved in their economic assessment.

4. Failures in Optimally Conserving and Managing Ecosystem Services

The operations of most economies depend to a significant extent on decisions by individuals or by relatively small groups of individuals. To a large extent, they make decisions (about the use of resources they control) in their own economic self-interest and do not take account of the economic consequences of their decisions for others, unless forced to do so by the law. As a result, the conservation and management of ecosystems (or more generally, the way in which the biosphere is utilized) is less than ideal from an economics point of view in the sense that it adds to economic scarcity rather than reducing it for society as a whole. The social economic loss can be quite large and the system can result in decisions about resource-use which result in unsustainable development.

For example, landholders and prospective landholders are mainly interested in the economic benefit they themselves can gain from the possession and use of land. These are usually benefits obtained on site. They ignore off-site benefits and costs which impact on others for which they do not receive any payment or incur any cost. These off-site effects usually result in the misallocation of resources and involve environmental externalities or spillovers or the attributes of public goods or bads (see, for example, Tisdell, 2005, Ch. 3; 2009, Ch. 3). Such effects are known to be sources of market failures in market systems but they can also result in resource misallocations in other economic systems as well.

This type of failure is also liable to generate political failures as well. Those individuals who expect to gain considerably from being allowed to carry out economic developments have an economic incentive to lobby politicians and public administrators to allow such developments even though they may have adverse environmental spillovers. Those who are adversely affected may not find it worthwhile to mount a counter political campaign even if their anticipated losses in total exceed the economic benefits to the developers. This can occur, for instance, if many are adversely affected by a small amount by environmental spillovers. In such cases, the transaction costs involved in mounting political opposition is a barrier to political lobbying by opponents.

Also the power to approve a development in some cases is devolved to local government. Local governments will be inclined to approve a development when it benefits the local area but has adverse consequences for other local government areas. Central governments can find it difficult to counteract such tendencies. Furthermore, different government departments (ministries) have different missions and 'client' groups. Within government, they tend to support their 'client' groups. This can result in projects being approved that benefit a particular client group but which on balance reduce social welfare.

The problem can be illustrated but some simple examples. Consider an existing area of land or ecosystem. In an initial situation, I, the area might consist mainly of a natural or a near natural ecosystem. In this situation, the total private economic gains to those within the land area might be as shown by rectangle ABEF and the total net spillover economic benefits might be as shown by rectangle ECDF in Figure 1 (left hand diagram). Now suppose that expansion of the private use of this land area (for example, for agriculture) is allowed. The economic benefits of those using the area now increase to the equivalent of the area of rectangle A′B′E′F′ but net spillover benefits decline to E′C′D′F′. Total economic benefits also decline because the area of rectangle A′B′C′D′ is less than that of ABCD. Consequently, permitting development of this area reduces economic welfare.

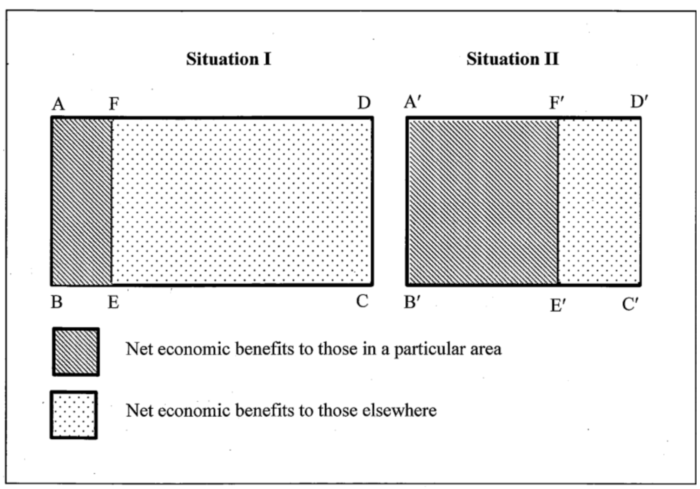

Figure 1. A case in which economic development in a particular area adds to the economic benefit of those in the area but reduces net economic benefits elsewhere and lowers total economic benefit.

While in the case illustrated in Figure 1, social economic welfare decreases as a result of changed land-use, this is not always the case. For example, the following are possible:

1. The development of a local area to increase provisioning services may reduce economic benefits obtained by other areas but the gain in local economic benefits may exceed the loss to other areas. In this case, total economic benefits rise.
2. The development of an area may raise local economic benefits as well as the spillover benefits to other areas. It has been suggested that the cultivation of rice in Japan has had this effect.

Consequently, it can be seen that a variety of economic outcomes are possible as a result of alterations in ecosystem or biosphere use. However, economic and political systems often do not favor optimal economic outcomes. For example, they may favor situation II in Figure 1 rather than situation I, even though the former reduces social welfare.

In assessing the alternative possibilities for ecosystem or biosphere use, the question also arises of deciding on the size of the geographical area for which spillovers are to be taken into account. To what extent should they be taken into account at the provincial, the national or global level. Many national governments are reluctant to take global spillovers into account in deciding on resource use because they act only in the national interest.

5. Difficulties in Determining the Economic Value of Ecosystems and Alterations to These

Possibly the greatest challenge faced in the economic valuation of ecosystems and alterations to these is to obtain relevant reasonably accurate biophysical data and reliable estimates of economic values (Norgaard, 2010). Economic evaluations of alternative uses of the biosphere or changes in ecosystems will be faulty if the relevant estimates of the ecosystem services or their economic values are quite inaccurate. The high cost of obtaining reliable estimates of these items is a major obstacle to the accurate economic evaluation of ecosystems and possible alterations to these.

An additional difficulty is that often the biophysical effects of changes to land-use of ecosystems are area specific and are difficult to model in advances of these changes. Often the effects can only be determined after the changes are made. Secondly, many of the methods for determining economic values are known to have significant limitations, especially for commodities that are un-marketed. This is true both of stated preference methods (such as the contingent valuation method) and revealed preference methods, for example, the travel cost method. Furthermore, data collection for the application of these methods can be quite costly. Despite the fact that there are two main sources of inaccuracies to contend with in evaluating alternative uses of the biosphere or of ecosystems, there will still be cases where the biophysical and economic evidence is sufficiently compelling to show that one form or pattern of use of the biosphere or an ecosystem is preferable to another or others. Even with uncertainty and bounded rationality, improved (or even optimal) choices about the use of resources can still be possible (see, for example, Tisdell, 2009, pp. 143-145).

Because of the high costs of the economic valuation of ecosystems and of potential changes to these systems, the question arises of who will pay for these valuation studies. Those who want to change an ecosystem for their commercial gain are unlikely to support such studies if they feel that such studies will not support their plans. They may also influence governments not to support such studies. Again, bias may occur in some valuation estimates. For example, estimates by the Research Council of Japan (2001) of the value of ecosystem services generated by Japanese agriculture could be on the high side. The study does not account for the ecosystem services which could be generated by alternative uses of agricultural land (or by some of it) by forestry, for instance.

Because of the high costs involved in valuing ecosystem services, various techniques are employed to reduce these costs. For instance, benefit-transfer analysis is frequently used. Net benefits that have been estimated for an ecosystem or type of land use in one location may be assumed to apply in another location. However, one has to be sure that the situation in a location where data are unavailable is similar to that where data are available. As pointed out by Pagiola *et al*. (2004, p.12, Box 3.2), situations which may seem similar often turn out to be different.

Furthermore, several studies rely on the cost of replacement method to estimate the economic value of ecosystem services and do not estimate the economic demand for these services. This reduces the amount of information needed for the economic valuation but it still can be quite demanding of information. If this demand for replacing ecosystem services which are liable to be lost as a result of changes to an ecosystem, exceeds (or nearly exceeds) the cost of replacing these, this method is sound. However, if this is not the case, this method overstates the economic value of the ecosystem system services that are liable to be lost if the ecosystem is altered (see, for example, Tisdell, 2012b). Pagiola *et al*. (2004, pp. 11-12) are highly critical of this method. Nevertheless, it can be useful in some cases and economic methods for estimating demand can also have significant limitations, especially when ecosystem services are not marketed.

In applying the cost of replacement method, the following should be kept in mind.

1. Is it possible to replace the ecosystem services by human investment or intervention? In some cases, it may be impossible and/or the replacement may be an imperfect substitute for the ecosystem service under consideration.
2. If more than one form of replacement of an ecosystem service is possible, then the least cost alternative should be chosen.
3. Determining costs of replacement are not straightforward. Engineering and specialist advice is likely to be needed to estimate such costs, and it may be difficult to get accurate estimates.

6. Payments for Supplying Ecosystem Services

One method which has been suggested to ensure the supply of ecosystem services is to pay landholders for adopting actions to secure the supply of these services, for example, paying landholders to plant trees or to refrain from tree removal. While this can sometimes be justified and effective, payment for ecosystem services (PES) is just one possible environmental policy instrument for controlling the supply of ecosystem services. Whether it is the appropriate instrument to use depends on the circumstances. Furthermore, the possible limitations of PES should be kept in mind. Let us consider these aspects taking tree cover as an illustration.

On properties that already have tree cover, charges (fees or taxes) may be imposed on the removal of trees, or this may be prohibited, except in special circumstances. An alternative is pay landholders for the retention of existing tree cover. Whereas the first approach imposes an economic burden on landholders and reduces their potential income, the second method imposes an economic burden on those who pay landholders for tree retention. In some cases, those who pay are taxpayers. PES tends to increase the income of landholders whereas taxes or restrictions on tree removal tend to reduce their income. The different policies have different consequences for the distribution of income.

If it is intended to increase the tree cover on existing properties, there seems little alternative but to pay landholders to do this. Then the question arises of how much should they be paid. Should they be just enough to induce them to plant and take care of the required number of extra trees or should they be paid more? For example, should they be paid the total economic value of the ecosystem services which they provide? A problem is that the economic value of the services provided is often poorly known. Moreover, this value depends to some extent on the combined planting of trees by all landholders. Consequently, the contribution to ecosystem services collectively is greater than the sum of that of each individual landholder.

Instituting policies to ensure the supply of ecosystem services involves agency costs and these need to be taken into account in assessing the economics of intervention. Bodies responsible for administering these policies must monitor the behavior of all landholders subject to these environmental policies. Where the policies are instituted by a central government, this usually requires power to be devolved to local bodies (governments) to ensure that these policies are complied with. However, the devolution of power also provides scope for local governments to deviate from the policy of the central government to some extent because of managerial slippage.

A further problem is ensuring the availability of finance to enable continuity of payments for ecosystem services (Pagiola, *et al.*, 2004). There is no guarantee that donors and governments will maintain payments for ecosystem services. Therefore, payments for the provision of ecosystem services may end abruptly and the supply of these services may be prematurely reduced.

A positive contribution of payments for ecosystem services is that they often assist those in poverty. In China for example, farmers in mountainous areas are usually relatively poor. They have benefited from China’s Grain-for-Green Program discussed in the next section.

The Common Agricultural Policy (CAP) of the European Union has also been altered to provide support for farmers based in the multifunctional effects of their activities including their supply of ecosystem services (Tisdell and Hartley, 2008, pp. 76-80). This means that subsidies to farmers in the EU no longer depend only on their volume of output but also take into account environmental and social considerations. Compared to earlier agricultural policies, this policy appears to add to social economic welfare in Europe. It reduces the likelihood that agricultural supplies will exceed demand. This was a problem in Europe in the past.

7. Restoring Ecosystem Services in China

China has embarked on the largest tree planting program of any country with the intention of restoring or supplying particular ecosystem services. It has five key afforestation projects managed by the State Forest Administration. Of these programs, the Grain for Green Project accounted for the largest area planted in the period 2001-2010, followed closely by the Three Norths Shelter Forest System Project (Phase IV) and then the Forest Industrial Base Development Program. The five projects are listed in Table 2 together with their purpose and planted areas in the period 2001-2010. In addition, there is a Wildlife Conservation and Nature Reserves Development Program but its main focus appears to be on retaining natural vegetation rather than tree planting.

Further information about the projects listed in Table 2, including their geographical spread is available in Cao, Chen, *et al.* (2011). It can be deduced from Table 2 that efforts to combat desertification accounted for the largest area of tree planting in the period 2001-2010. The Three Norths Shelter Project plus the Sand Control Program accounted for the afforestation of 32.7 million ha. and the Grain for Green Project (which applies to steeply sloping land) also included (includes) some parts of China subject to desertification.

There has been a significant increase in forest cover in China since 2000 and there are plans to increase forest cover in China to 26% by 2050 (Wang, *et al.*, 2007) from the present level of around 19%. However, there is debate about just how much of the increase in forest cover is due to afforestation projects (Cao, Chen, *et al.*, 2011, p. 241). In some areas, bans on grazing have been a major factor in increasing tree cover as has been reduced cultivation of land (Cao, *et al.*, 2009). In fact, in some of China's arid and semi-arid areas tree planting programs have for ecological reasons resulted in vegetative changes that have increased soil erosion and desertification (Cao, 2008; Cao, Chen, *et al.*, 2011; Cao, *et al.*, 2009; Cao, *et al.*, 2010). Because these projects are the responsibility of the State Forest Administration, its mission may be biased in favour of tree planting rather than the restoration of grasslands. Restoration of grasslands appears to be a more effective strategy for combating desertification in arid and semi-arid areas of China than tree-planting (Cao, Sun, *et al.*, 2011).

Table 2. Major afforestation projects in China: their purpose and planted areas in the period 2001-2010

Project	Purpose	Planted area (million ha.)
Grain for Green Project	Soil and water conservation	32.0
Three Norths Shelter Forest System Project (Phase IV)	Desertification control	27.5
Forest Industrial Base Development Program	Wood production	13.3[(a)]
Sand Control Program	Desertification control	5.2
National Forest Conservation Program	Soil and water conservation	4.4

Note: (a) This is for the period 200-2015
Source: Based on Cao, Chen, *et al.* (2011), Table 1, p. 241

The question has also been raised of whether China's projects for restoring ecosystem services are benefiting the rural poor. Uchida *et al.* (2007) found that China's Grain for Green Program had been moderately successful in alleviating rural poverty amongst participants and that overall it had greatly reduced soil erosion. However, its long-term consequences for poverty alleviation were unclear because few beneficiaries had shifted to off-farm work (Uchida, *et al.*, 2007, p.617) and they tended to use subsidies to increase their numbers of livestock. Nevertheless, Zheng *et al.* (2011) report that the Grain for Green program compensation payments often failed to be made or have only been partially paid, with grain payments often being of poor quality. They state that "on average, only about 49 per cent of the grain subsidy and 23 per cent of the cash subsidy has been received by program participants" (Zheng, *et al.*, 2011, p. 3). Reasons include "local government reallocation of subsidies for other uses". This is an example of the agency problem mentioned earlier in this paper: local government bodies have some leeway in distributing funds because the central government cannot exercise complete managerial control over them.

Another possible problem is that environmental restoration may not be permanent. For example, Cao, *et al*. (2009, p. 1182) found from a survey in northern Shaanxi Province that 37.2% of their respondents participating in the Grain for Green Project "planned to return to cultivating forested areas and grassland once the project's subsidies end in 2018". The sustainability of environmental achievements in the long-term requires beneficiaries from payments for the provision of ecosystem services to find alternative sources of income that do not require them to revert to cropping and grazing the land on which trees and grassland have been restored. This may require payments and schemes to facilitate the movement of farmers to off-farm employment. However, the returns from off-farm employment in areas where poor farmers are located tend to be low. Migration to other areas may be required. If this is not feasible for older farmers, younger members of farm families may be encouraged to migrate to urban areas, so as not to perpetuate poverty and its associated environmental problems.

8. Conclusion

The Millennium Ecosystem Assessment (2005) highlighted the importance for human well-being of ecosystem services but it could have been better balanced. It tended to focus on the positive benefits provided by ecosystems and ignored the disbenefits generated by some ecosystems. Furthermore, it provided little concrete policy advice and did not estimate the economic value of different ecosystems.

Economists have provided economic estimates of the value of different ecosystems globally as well in particular local areas. Global estimates (for example, by Costanza, *et al.*, 1997; de Groot, *et al.*, 2012) appear to be more an exercise in creating awareness of the value of different ecosystem services (many of which are outside the market system) rather than providing practical policy advice on which should be given priority for conservation and to what extent. It is easier to provide practical advice about such matters when the economic valuation of more localized supply of ecosystem services is undertaken, as was illustrated by the study by Xue and Tisdell (2001).

It was shown that significant failures occur in optimally conserving and managing ecosystem services. There is because market failure and political failure and these frequently go hand in hand. Furthermore, several difficulties are encountered in accurately valuing ecosystem services and changes in the value which are likely to arise from altering an ecosystem. Both relevant biophysical and economic data are often lacking and it can be very costly to improve the quality of this data. There is a bounded rationality problem.

Payments for supplying ecosystem services are often recommended as a means for regulating the supply of ecosystem services. In some cases, however, alternative policy instruments should be considered. Some of the advantages and disadvantages of paying for the supply of environmental were outlined.

China has embarked on a major program to restore ecosystem services by means of its afforestation projects. These projects (in conjunction with other factors) have contributed to a substantial increase in tree cover in China which it is planned to increase even further. Nevertheless, there are still some difficulties to address. These include the inappropriateness for control of desertification by tree planting in some arid and some semi arid areas, the use by local governments of centrally provided funds intended for tree planting for other purposes, and the possibility that vegetation restoration will not be permanent because reversion is likely to some extent when subsidies cease. Therefore, in the long-term, it is necessary to consider the scope for off-farm employment and migration of farm families.

Despite all these difficulties, there is no doubt that China has made progress in taking into account the ecosystem services provided by different forms of land use. For example, the State Forest Administration now does not envisage its sole task to be to the supply wood. It has adopted a wider mandate. Nevertheless, this body seems to be over focused on tree-planting, compared to other forms of re-vegetation, as a means to supply the ecosystem services that it wants to provide. Of course, there is also a need for China to pay more attention to the environmental impacts of other ecosystems, such as its agroecosystem. The task is a major one for China. Most countries face similar issues.

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