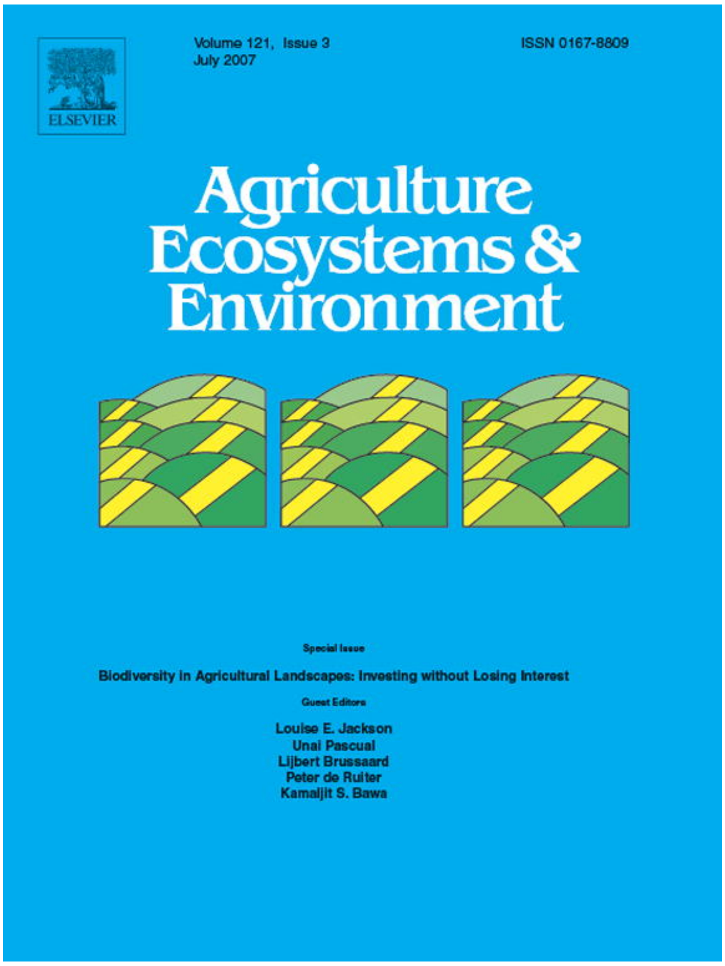
Volume 121, Issue 3
July 2007
ISSN 0167-8809
ELSEVIER
Agriculture Ecosystems & Environment
Special Issue
Biodiversity in Agricultural Landscapes: Investing without Losing Interest
Guest Editors
Louise E. Jackson
Unai Pascual
Lijbert Brussaard
Peter de Ruiter
Kamaljit S. Bawa

Available online at www.sciencedirect.com
ScienceDirect
Agriculture, Ecosystems and Environment 121 (2007) 256–268
Agriculture Ecosystems & Environment
www.elsevier.com/locate/agee

Developing incentives and economic mechanisms for *in situ* biodiversity conservation in agricultural landscapes

U. Pascual [a,*], C. Perrings [b]

[a] *Department of Land Economy, University of Cambridge, 19 Silver Street, Cambridge CB39EP, UK*
[b] *Global Institute of Sustainability, Arizona State University, Tempe, Box 873211, AZ 85287-3221, USA*

Available online 30 January 2007

Abstract

The main focus of this paper is agrobiodiversity and its effects on agricultural production within agricultural landscapes. The interest is to shed light about the fundamental causes of agrobiodiversity loss by focusing upon the institutional or meso-economic environment that mediates farmers' decentralized decisions. Since the main causes of farmers' decisions to 'disinvest' in agrobiodiversity as an asset lie in the incentives offered by current markets and other institutions, the solution to the problem also lies in corrective institutional design. This paper discusses the institutional issues involved in establishing market-like mechanisms for agrobiodiversity conservation. Three steps are highlighted in such process: demonstration (valuation), capture and sharing of conservation benefits (mechanism design). This information is then used to examine the potential success of nascent market creation incentive mechanisms for biodiversity conservation, including: (i) payments/rewards for ecosystem services, (ii) direct compensation payments, (iii) land use development rights, and (iv) auctions for biodiversity conservation. The potential gains to society from their use with regard to agrobiodiversity conservation are discussed and some illustrative examples involving their application in different parts of the world are also described.

Keywords: Agrobiodiversity; Market creation; Institutional design; Meso-economy

1. Introduction

The most important anthropogenic cause of agrobiodiversity loss is rapid land use and land cover change (LUCC) and the subsequent transformation of habitats (MEA, 2005). In agricultural landscapes LUCC usually takes the form of land development. Most land development at the landscape level stems from the decentralized economic decisions of economic agents, including small scale farmers, agribusiness and governments at different scales. The ecological causes and effects of such landscape transformations are increasingly well understood and documented, especially with regard to deforestation and desertification in developing regions (Lambin et al., 2001; Perrings and Gadgil, 2003). In agricultural landscapes, one impact of LUCC that is attracting increasing attention is the alteration of the flow of ecosystem services that are mediated by biodiversity (MEA, 2005; Perrings et al., 2006). This has significant implications for biodiversity conservation strategies in agroecosystems.

Agrobiodiversity is not a fixed asset that every person experiences similarly. Since it is experienced contextually, it is socially constructed (Rodríguez et al., 2006). There are differences in the way that social groups identify and value biodiversity-based services. Nevertheless, agrobiodiversity change can be seen as an investment/disinvestment decision made in the context of a certain set of preferences, 'value systems', moral structures, endowments, information, technological possibilities, and social, cultural and institutional conditions. An important starting point for science is therefore to understand how (a) biodiversity supports the production of ecosystem services, and (b) those services are valued by different social groups.

From an economic perspective, biodiversity change is most obviously a problem wherever it yields negative net benefits. More generally, it is a problem wherever it is

* Corresponding author. Tel.: +44 1223 337151; fax: +44 1223 337130.
E-mail address: up211@cam.ac.uk (U. Pascual).

0167-8809/$ – see front matter © 2006 Elsevier B.V. All rights reserved.
doi:10.1016/j.agee.2006.12.025

socially inefficient (given social distributional priorities). In most cases, this reflects market failures that are due to the existence of externalities (incomplete property rights) and the public-good nature of conservation. That is, there exists a wedge between individual agents' perceived net benefits from LUCC actions and those realized by the community that is affected by those same actions (Swanson, 1998; Perrings, 2001; MEA, 2005). Part of the problem in understanding the social value of biodiversity change is that while some of the opportunity costs of conservation or foregone benefits from land development are easily identified, there remain important gaps in the understanding both the on- and off-farm benefits of agrobiodiversity conservation.

In many cases a preservation-centred strategy that involves allocating valuable resources (e.g. land) towards maximum *in situ* biodiversity conservation will not be socially efficient. The cost, in terms of the foregone food and fibre production, of allocating an additional hectare of land for conservation, may be larger than the additional conservation benefits. The 'optimal' intensification debate reflects this fact (Green et al., 2005). Such a debate would be enriched if scientists were able to identify the complex relationships between land management options, biodiversity impacts, changes in ecological services and their values (Perrings et al., 2006).

LUCC and concomitant agrobiodiversity effects depend on the social, economic and institutional conditions that frame economic agents' decisions. In this context, institutions encompass formal rules (e.g. laws, constitutions) and informal constraints (norms of behavior, self-imposed codes of conduct) that govern land users' behavior. They can also be referred to as 'rules in use' (North, 1990) as the ones found in markets. In this vein, decentralized decisions regarding the desired level of *in situ* planned agrobiodiversity, e.g., crop and livestock genetic diversity (Vandermeer and Perfecto, 1995; Jackson et al., 2007) usually depend on conditions in the relevant food, fuel and fiber markets (Smale et al., 2001). Market signals affect farmers' private land use decisions by fixing the private net benefits of their individual actions, given their risk aversion and rate of time preference.

One type of agrobiodiversity that is reasonably well-understood is genetic diversity of cultivars and breeds (Smale et al., 2001). Since the social insurance benefits of higher levels of crop genetic diversity are not rewarded in many current markets, farmers have little private incentive to conserve genetic diversity (Perrings, 2001). The most profitable decision is frequently to grow only a few crop varieties, and not to invest in conservation of the varieties that are less 'favored' by the market.

The problem, in this case, lies both in the public good nature of conservation, and the fact that there are no markets for off-site ecosystem services that depend on on-farm agrobiodiversity. A good is catalogued as public if it does not exhibit rivalry and excludability characteristics. Biodiversity is non-rival as one individual's use of biodiversity does not affect another individual's use of it, i.e., individuals can be equally satisfied simultaneously by the fact that biodiversity is conserved. It is generally non-excludable because it is impossible or very difficult to exclude or prevent someone from benefiting from its conservation. In the case of genetic diversity, farmers who maintain *in situ* crop genetic diversity are essentially conserving a global public good and thus they can be seen as net-subsidizers of modern agriculture and food consumers worldwide. However, global institutions are not in place to provide compensation for generating such global benefits. Indeed, one reason for the profitability of modern specialized agriculture is that it is free-riding on those farmers who are investing in such genetic diversity. The net result is that global crop genetic diversity is being rapidly reduced, since the custodians of the global genetic portfolio are uncompensated by current international markets, and there are no corrective policies or mechanisms in place. For other types of agrobiodiversity, e.g., at the community and landscape level, the situation is even more complex because inventories and functions are so much more difficult to assess.

The fundamental causes of agrobiodiversity loss, therefore, lie in the institutional or meso-economic environment that mediates farmers' decentralized decisions. This paper discusses such institutional (meso-economic) dimensions of *in situ* agrobiodiversity change in the context of a framework that identifies: (i) the forces at play at the microeconomic (farm economy) and meso-economic (market/institutional) level leading to (dis)investment in biodiversity within agricultural landscapes, and (ii) the economic consequences of biodiversity change at the individual and social level. This allows us to discuss mechanisms that can help align the social and private values of biodiversity conservation.

The main focus of this paper is agrobiodiversity and its effects on the multiple services that agriculture provides to society, especially those related to the provision of foods and fibers within agricultural landscapes. The impacts of agriculture on wild species without apparent agricultural value, their habitats, and their contribution to other non-agriculturally related ecosystem services are not emphasized. The scope is purposefully limited, and the paper is organized as follows: The next section addresses institutional failures at the micro-meso-and macro-scales. In Section 3 we discuss the private and social value of agrobiodiversity conservation. Section 4 then addresses the two main stages in market creation: capture and sharing of conservation benefits. We consider various nascent and potentially fruitful incentive mechanisms that can re-create decentralized markets to foster agrobiodiversity conservation. A final section recapitulates the main points and draws out the implications for the conservation of agrobiodiversity.

2. The drivers of agrobiodiversity change

Farmers' agrobiodiversity choices reflect a number of factors aside from market prices, including the social, political and cultural conditions in which they operate. These are generally exogenous to the farmers' own decisions (Lambin et al., 2001), but are strongly influenced by policy at the national and international levels. The problem we consider is the interaction between micro-economic (decentralized) farmers' decisions and meso- and macro-economic/institutional factors.

At the micro-scale, the household, family farm or agri-business constitutes an institution itself with its own behavioural 'rules' that impinge on LUCC decisions. In the case of farm households if the internal rules are such that there is intra-household gender discrimination, the species to be conserved may be determined by gender dominance. In many African drylands, for example, women favor planting for fuelwood and men for fruit trees, because it is the women who tend to collect fuelwood, while men control cash income generated by selling fruit in the market. This helps to explain why, even as the sources of fuelwood continue to recede in many African countries, fruit trees are often planted (Dasgupta, 2000). This is an example of institutional failure at the household level.

At the macroeconomic level, institutional or policy failures are often more evident and their effects more far-reaching. Macro-economic institutions include both national and international policies. Many of these affect the incentives facing individual farmers. One clear example of institutional failure at the macroeconomic level lie in the perverse agricultural production subsidies, tax breaks and price controls that not only make a biodiversity-based agriculture uncompetitive, but that have systematically distorted farm-level decisions in both developed and developing countries for decades (Tilman et al., 2002). At the beginning of the century, subsidies paid to the agricultural sectors of OECD countries averaged over US$ 324 billion annually (about one third the global value of agricultural products in 2000) (Pearce, 1999).

Consider the following illustrative examples from Sudan (Barbier, 2000) and Indonesia (Tomich et al., 2001). Barbier (2000) analyzed the impact of distortionary macroeconomic price policies affecting the 'gum arabic' (*Acacia senegal*) agroforestry system in Sudan. It is planted in bush-fallow rotation and intercropping farming systems. The gum produced by the tree is traditionally exported for manufacturing industries. Additionally, *A. senegal* provides ecological services such as the provision of fodder for livestock, fuelwood and it offers an important regulatory ecological function against desertification, as it serves as a wind break for dune fixation. Indeed, given the potentially high financial returns to the gum arabic coupled with its important environmental benefits, this land use system seem to be ideal in arid regions. But as Barbier (2000) notes, in recent decades, macroeconomic policies by the Sudanese government, largely based on distortionary (overvalued) exchange rates and export policies, e.g., high export taxes, have meant that the rate of return to farmers for producing gum arabic has declined relative to its alternative competitive annual cash-crops, i.e., sesame and groundnuts, and even to staple crops such as sorghum and millet. This is a compelling reason for farmers to disinvest in gum arabic stands in agroforests.

Tomich et al. (2001) reported that research into rubber agroforestry systems shows that extensively managed agroforests provide greater biodiversity benefits than intensive rubber tree plantations, but that at the current real producer price of rubber, relative to the minimum wage rate, returns to farm labor are 70% higher in intensive plantation systems than agroforestry. Once distortionary prices, including tax and subsides for rubber production, are eliminated, however, labor returns to rubber production in extensive agroforestry systems outweigh its alternative plantation returns by 30%.

Other important macro-level institutions that affect both micro-and meso-economic institutional contexts include the intergovernmental organizations (World Bank, International Monetary Fund, United Nations Development Programme) and international agreements (the General Agreement on Tariffs and Trade, the Sanitary and Phytosanitary Agreement, and the International Plant protection Convention). In some cases, they affect agrobiodiversity by limiting the choice of management strategy or technology used by farmers. In others, they work by encouraging the diffusion of new technologies or by dispersing new crop varieties, bio-control agents, pests and pathogens (Perrings, 2005). As in the case of direct subsidies, these indirect influences on farmers' decisions change the private returns on farm investments, often in ways that discourage agrobiodiversity conservation. Amongst other effects of the incentives offered directly and indirectly by such institutions are the loss of forest and wetland habitat, the devegetation of watersheds, the loss of soil and aquatic biodiversity through the application of pesticides, nitrogen and phosphorous, the depletion of many beneficial pollinators and pest predators (Scherr and McNeely, in press), and the introduction of invasive species (Mooney et al., 2005).

The solution is to 'fix' these incentives – to realign the mismatch between the private interests of farmers and those of society at large – although markets do not operate in a vacuum. Their operation relies on other supporting institutions including those that shape the regulatory environment. Hence, correcting for market failures is a necessary but not sufficient condition for readdressing agrobiodiversity loss. Investing in adequate (effective, stable and resilient) institutions that allow markets to operate is also necessary to create favourable conditions that can lead farmers to further invest in biodiversity conservation in a decentralized and voluntary fashion.

An additional problem is that biodiversity is a public good, and as with other public goods, will be underprovided if left to the market. Even if relative prices were fixed to reflect the social opportunity cost of biodiversity, there would still be an incentive to free ride on the conservation efforts of others. Nevertheless, it is clear that correcting many of the perverse incentives facing farmers requires that the policy maker understands the value of agrobiodiversity. It is important, therefore, to link the process of valuation with the creation of new effective and efficient institutions for conservation. At the same time, it is important that the valuation of biodiversity is linked to the delivery of appropriate incentives to farmers. For example, the benefits to peasant households from conserving off-farm agrobiodiversity in forest margins needs to cover the costs in terms of foregone timber extraction revenues or the income that could accrue by converting such forest land to agricultural production for food security.

Economic valuation and the development of markets for biodiversity are potentially effective providing that they achieve (i) demonstration, (ii) capture, and (iii) sharing of biodiversity benefits especially taking into account the communities that face the opportunity costs of conservation (OECD, 2005). Demonstration refers to the identification and measurement of biodiversity values as the benefits from conserving it may not always be evident. It is the exercise of identifying the valuation pathways. This is a non-trivial task and much research is still needed (Opschoor, 1999; Jackson et al., 2007).

Capture, in turn, is the process of appropriating the demonstrated and measured biodiversity values in order to provide incentives for its conservation. This is achieved by regulations and markets to allow for such values to be made explicit and channelled from the beneficiaries (society as demander) to those who bear the cost of conservation (farmers as suppliers). For example, a niche market for ‘biodiversity-friendly’ products would channel the revenues to those farmers that certify the production of such ‘green’ outputs in order to compensate them for the foregone higher earnings from a privately more rewarding alternative land-use. The market, in this case, may internalise the biodiversity values through price premiums creating positive incentives towards biodiversity conservation decisions.

Lastly, effectiveness ultimately depends on whether the benefits of the provision of the public good (conservation of biodiversity) are distributed to those who ultimately bear the costs of conservation. Following the above example, the price premium of the certified biodiversity-friendly products would need to be channelled back to the producers. This is not a trivial task, as often a disproportionate part of price premium can be off-channelled to traders and middlemen (Bacon, 2005). At a global level, another example is that of the free-prior consent and benefit sharing agreement clauses imposed by the UN Convention of Biological Diversity with regard to bioprospection endeavours regarding plant genetic resources (ten Kate and Laird, 1999), This necessitates to effectively assert the property of bioresources and genetic resources in particular to the source country (c.f. UNCBD Article 15: Access to Genetic Resources).

3. Understanding the social value of agrobiodiversity

To demonstrate the value of agrobiodiversity, science can assist in (i) assessing the functional role of species in their crop- and non-crop habitats, (ii) identifying the biotic and abiotic components of agroecosystem structures that support the provision of ecological services at the landscape level and, (iii) assessing the contribution of such ecological functions to human wellbeing. The challenge is to translate such ecological interdependencies into tangible ecological services that can be valued from an anthropocentric perspective (Perrings et al., 2006). Here we address some of these complex issues by providing a conceptual framework of the links between agrobiodiversity as a stock (S), the provision of flows of ecosystem services (F), and the ‘total economic value’ (V) that this generates to society.

Fig. 1 illustrates such linkages in stylized way. It also shows the links between values, wellbeing at both individual and social levels. Since existing markets fail to align the social and private values of agrobiodiversity through LUCC, policies are needed to correct for such market failure. A feedback loop exists between policies, LUCC and agrobiodiversity at the landscape level. The dotted arrows represent links that are difficult to appreciate and that need to be further investigated.

3.1. The direct ‘instrumental value’ of agrobiodiversity

Managed on-farm biodiversity can be represented as a stock or economic asset (S1). The asset represents the mix of species and communities that supply a flow of ecological services on-farm (F1) that can directly benefit farmers by maintaining or enhancing agricultural productivity. This is achieved, for example, by the control of on-farm *destructive biota*, such as weeds, insect pests and microbial pathogens (Swift and Anderson, 1993).

When on-farm biodiversity supports the productivity of crops by enhancing yields or substituting for the use of purchased capital inputs, such as pesticides, such biodiversity has an instrumental or ‘use-value’ for farmers (V1). Usually, V1 is more apparent and relatively more important in small-scale farming in resource-poor areas where access to capital inputs (e.g. irrigation and agrochemicals) is constrained, and where biodiversity is often managed to regulate pest and diseases, soil formation and nutrient recycling (Altieri, 1999). An example is that of the meso-American shifting cultivation ‘milpa’ system in which maize/squash/bean polycultures are more stable than monocultures (Altieri, 1999). This is reflected in the S1∼F1∼V1 link in Fig. 1. If farmers are able to conserve such biodiversity, and if this permits them to stabilize and enhance agricultural income (V1), then this strategy can be viewed as sustainable (Conway, 1993).

Different crop mixes at the plot level and the diversity of uncoordinated individual agricultural management strategies creates a mosaic of agrobiodiversity at the landscape

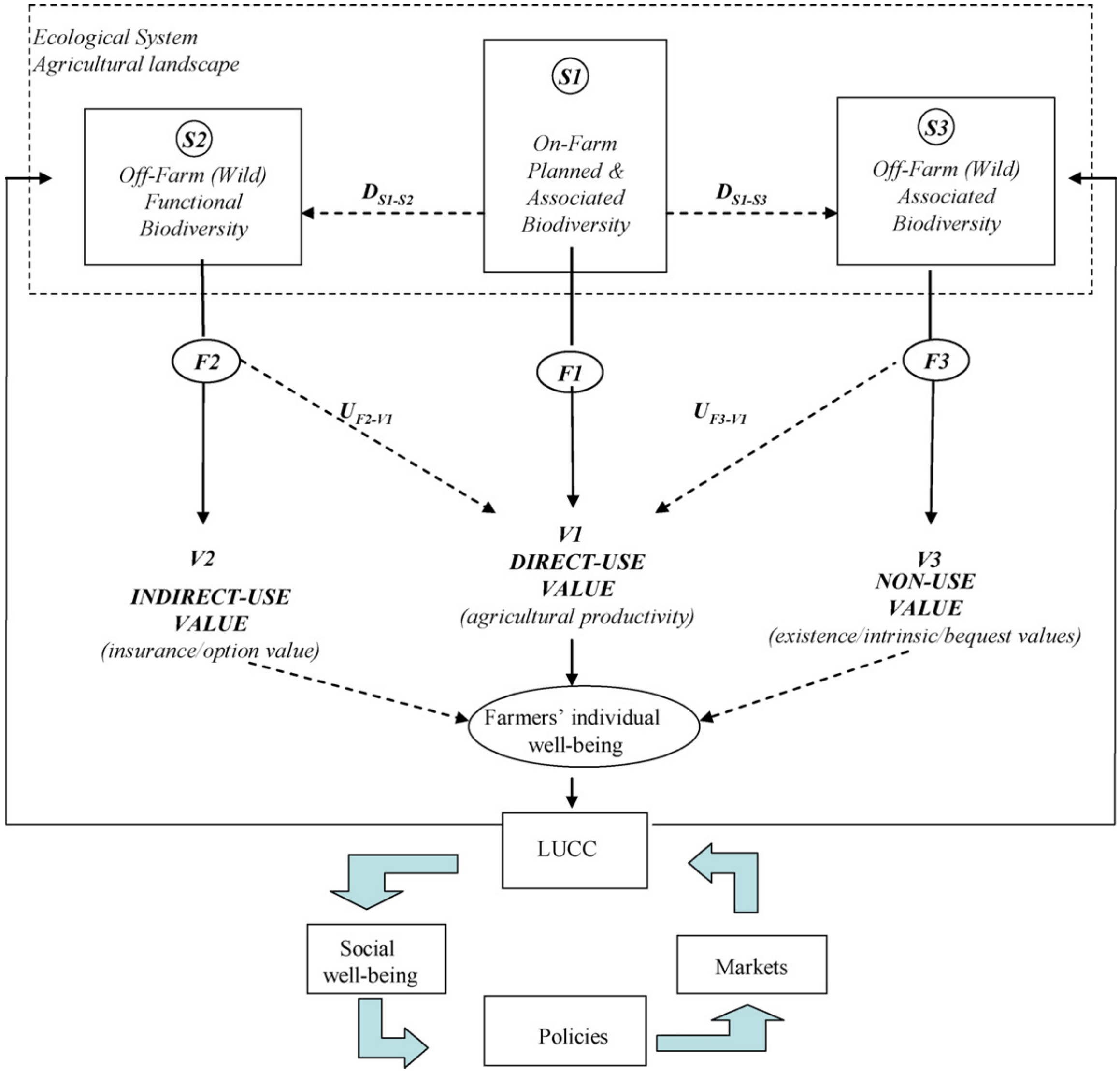

Fig. 1. A stylized framework of the linkages between biodiversity levels (stocks, S), flows of ecological services (F) and economic values (V) in agricultural landscapes leading to LUCC and policies that aim at aligning the private and social values of agrobiodiversity. The ecological system governing the interaction between on- and off-farm biodiversity stocks within agricultural landscapes provides flow of ecological services that benefits individual land users and society as a whole in different ways. Individual land users compare the directly perceived benefits of conservation and the opportunity costs to decide on their privately (decentralized) optimal land use and the level of (dis)investment in biodiversity. This in turn affects social wellbeing and policies are sought to change such perceived net benefits.

level. In this process, there are effects of changes in on-farm planned biodiversity (S1) on off-farm functional diversity (S2) at the landscape level. For example, the amalgamation of agricultural fields tend to produce homogeneous farmed landscapes leaving only a fragmented non-crop habitat that affects both the off-farm functional (S2) and associated (S3) diversity (Bélanger and Grenier, 2002; Benton et al., 2002; Tscharntke et al., 2005). We refer to this as a downward (or forward) biodiversity effect that links decentralized farmers' decisions and landscape level agrobiodiversity. This relationship is depicted in Fig. 1 with the dotted arrows D_{S1-S2} and by D_{S1-S3}. The ecological-economic problem is to identify the mosaic of connected habitats that best supports both farm production (F1) and its value to farmers (V1) and the supply of off-farm ecosystem services (F2 and F3) that support off-farm values (V2 and V3).

There are also upstream (or backward) biodiversity effects. There is increasing evidence of the positive effect of off-farm biodiversity on on-farm productivity. Often this is associated with off-farm landscape level generalist species (S2) that provide pollination and biological control services against pests and invasive species. This is depicted by the dotted arrow $U_{F2\sim V1}$. In this case, the flow of ecological services provided by off-farm functional species (S2) generates an indirect use value to farmers—it can provide financial savings to farmers. For example, Kremen et al. (2002) show that more intensive agricultural land management relative to less intensive systems, such as organic farming, increases the cost of pollination to farmers. In another study, Ricketts et al. (2004) estimate the economic cost of the reduction of pollination services originating from off-farm forest habitats to coffee production in a Costa Rican farm to be in the order of US$ 60,000/ year. This would be an approximate figure as neither of these studies considers the increased income generated by converting the neighboring forest habitat to agriculture. Similarly, the loss of off-farm pollinators and pest-predators increases the cost to farmers of pest and disease control (Symondson et al., 2002). At the same time, habitat fragmentation increases the risk of invasion by unwanted destructive off-farm species at the landscape level (Östman et al., 2003; Perrings, 2005).

Finally, we should note that transboundary landscape effects also affect upstream linkages (depicted by the dotted arrow $U_{F3\sim V1}$). Off-farm biodiversity at regional and even global scales can affect the long run productivity of local agricultural systems. One well known example is the relationship between the diversity of insectivorous birds, some of which migrate from tropical forests in Latin America to Canadian boreal forests, and which help to regulate the productivity of forest stands by controlling the destructive population of spruce budworms (*Choristoneura fumiferana*) (Holling, 1988).

3.2. The indirect use value of agrobiodiversity: the insurance hypothesis

While economists have long been aware that biodiversity has an 'indirect' value through the provision of regulating ecosystem services (Barbier, 1989), there have been few attempts to estimate this value for particular systems. Within the present framework, possibly the most important value of off-farm functional diversity (S2) stems from its role as an insurance mechanism (F2) (Folke et al., 1996; Loreau et al., 2002; Baumgärtner, submitted).

Ecologists argue that over small scales (e.g. the crop-field level) an increase in on-farm species richness and the diversity of overlapping functional groups of species enhances the level of functional diversity, which, in turn, increases ecological stability (Tilman et al., 1996) and resilience (sensu Holling, 1988, 1996). In this sense, resilience refers to the size of perturbation that is required to transform a system from one state to a different state, and is frequently increasing in the number of species that are apparently 'redundant' under one set of environmental conditions, but that perform important functions under different environmental conditions (Holling, 1988; Peterson et al., 1998). Further, following Carpenter et al. (2001), resilience of an adaptative agro-ecosystem would be determined primarily by: (i) the amount of disturbance that the system can absorb and still remain within the same state or domain of attraction, (ii) the degree to which the system is capable of self-organization, *versus* the lack of organization, or organization forced by external factors, and (iii) the degree to which the system can build and increase the capacity for learning and adaptation.

For instance, in biodiversity-poor intensive agricultural systems that depend on increasing use of artificial inputs, the agricultural system can be locked into a narrow range of agricultural technologies. At one level this can make the system more stable in the sense that there is less variation in the producer's economic activities following minor perturbations, but conversely, it may also reduce the capacity of that system to absorb greater environmental or economic shocks, such as sudden and unexpected commodity price changes. By eliminating options towards productive diversification, a reduction in agrobiodiversity may also lock farmers into obsolete agricultural technologies (Perrings, 1998).

It follows that maintaining a wider portfolio of technological and natural resource-based options in agricultural systems is likely to maintain or enhance the capacity to respond to short-run shocks and stresses in constructive and creative ways. Various recent studies have analyzed the contribution of crop diversity to the mean and variance of agricultural yields and farm income (Smale et al., 1998; Schläpfer et al., 2002; Widawsky and Rozelle, 1998; Di Falco and Perrings, 2003, 2005; Birol et al., 2006). One main conjecture is that risk averse farmers use crop diversity in order to hedge their production and income risks, especially when affected by changing market conditions. Hence, off-farm biodiversity through its insurance mechanism (F2) can provide an important insurance value to farmers ($F2\sim V2$) and productivity enhancing services (this is a backward linkage, U_{F2-V1}). To the individual farmer, however, the insurance effect may not generally be enough to justify conservation when there is ample access to improved artificial capital inputs, e.g., fertilizers, improved seeds, etc. The insurance value is thus better perceived and exploited in agricultural landscapes that are mainly associated with agroforestry and agroecological production systems. In addition, the insurance value can be associated with the idea of 'option value', reflected in the important efforts to maintain *ex situ* genetic resource conservation (Jackson et al., 2007).

3.3. The infrastructure value of agrobiodiversity

Similarly, while there has been recognition of the value of biodiversity in underpinning ecosystem functioning and processes which is sometimes referred to as 'primary' (Turner and Pearce, 1993), 'infrastructure' (Costanza et al., 1997) or 'contributory' (Norton, 1986) value, there have been few attempts to estimate this. This is partly due to the difficulty of capturing the interaction between species, and more generally the functional links between on-and off-farm biodiversity.

Economists first modeled this by assigning species the status of 'intermediate inputs' (Crocker and Tschirhart, 1992) due to their role in supporting more directly other productivity-enhancing species. The same idea can be generalized to say that species have value deriving from their indirect role in the production of valuable goods and services that is conditional on the state of the environment. So, for example, the derived value of members of a functional group of species, each of which performs differently in different environmental conditions, will vary with those conditions. Species that appear to be redundant in some conditions, will still have value depending on the likelihood that the conditions in which they do have value will occur in the future (Loreau et al., 2002). This translates easily into the idea that the cost of species deletion becomes the cost of the alternative ways of securing the same productivity outcome, as long as those species contribute to the productivity of the agricultural ecosystem.

Lastly, it should also be pointed out that besides biodiversity's effect on productivity (F1) and stability/resilience (F2), associated off-farm biodiversity (S3) can also provide other benefits to society, e.g., cultural and recreational (F3). For instance, in industrialized countries where natural habitats are scarce, there are important landscape values of farmland (V3), that typically consist of the benefits derived from the scenic beauty generated by a rural landscape such as, open fields, orchards, and herds of livestock grazing in green meadows (OECD, 1993; Cobb et al., 1999). The implication of the realization of such values in the European Union, for example, has spurred renewed emphasis on the role of multifunctional agriculture to secure such recreational and non-instrumental social values and has provided impetus for the design and implementation of novel agri-environmental policies (Hodge, 2000).

4. From demonstration to capturing and sharing the benefits of agrobiodiversity conservation

There are compelling reasons to devise and implement incentive mechanisms for agrobiodiversity conservation. Incentives can be categorised into two main groups: (i) moral suasion, regulation and planning, e.g. by preventing specific land management practices or by designating conservation zones within agricultural landscapes, known as agroecological 'no take' zones resembling nature reserves and parks, and (ii) market creation for agrobiodiversity conservation given the power of decentralised land use decisions.

Market creation stems from a simple but powerful idea, i.e., markets can be devised to signal the opportunity cost to local land users of agricultural practices that affect agrobiodiversity either positively or negatively. Ideally, such incentives need to address the above mentioned forward and backward agrobiodiversity linkages and, thus, work at the landscape level. But this implies that such incentives may affect the livelihoods of large numbers of farmers. This adds a further layer of responsibility to public agencies to be aware of the distributional implications of alternative incentive measures.

Markets can take different forms. One is for interested 'buyers' such as firms and NGOs to purchase land use rights or permits. For instance once a logging permit is obtained, a conservation NGO may decide not to extract timber but instead to conserve the land for its biodiversity. More specifically, within agricultural landscapes 'use rights' include rights of access to particular biological resources, e.g., game, fish and non timber forest products, or other goods and services that may be associated with biodiversity, such as those associated with organic agricultural products.

Land use rights are currently being extended to enable voluntary contractual arrangements between farmers and off-farm users of ecosystem services that are affected by actual farm management. Here we discuss the potential of using markets in conjunction with land use rights for agrobiodiversity conservation at the landscape level focusing on various relatively nascent mechanisms that allow the capture and distribution of conservation values: (i) 'Payments/Rewards for Environmental Services', P(R)ES, (ii) Direct Compensation Payments, (iii) Transferable Development Rights (TDRs), and (iv) Auction Contracts for Conservation (ACCs).

4.1. Payments/rewards for environmental services

P(R)ES are voluntary transactions, not necessarily of a financial nature, in the form of compensation flows for a well-defined environmental service (ES), or land use likely to secure it. The notion of 'rewards' is used to acknowledge that transactions from beneficiaries to providers may not need to be based on a financial flow. It can also involve in-kind transactions that may include a myriad of valuable goods and services from the beneficiaries point of view, which can take intangible forms in diverse situations, such as knowledge transfer. P(R)ES is paid/rewarded by the beneficiaries and shared by the providers of the ES after eventually securing such compensation. The latter conditionality element frames such schemes under the 'Provider Gets Principle' (Hodge, 2000).

P(R)ES are often designed to address problems related to the decline in some environmental services, such as the provision of water, soil conservation and carbon sequestration by upland farmers who manage forest-lands in upper watersheds. In essence, such compensations are intended to internalize the positive externalities generated by upland farmers who can maintain the flow of valuable services that benefit lowland farmers or urban dwellers. However, a key obstacle in the successful implementation of PE(R)S arises at the value 'demonstration' stage, especially due to the scientific uncertainties underpinning the linkages between alternative land uses and the provision of the targeted environmental services.

Regarding the effectiveness of the capture and sharing of the benefits, recent evidence identifies various necessary conditions, including the need: (i) to clarify the level of excludability and rivalry of such ES by beneficiaries and providers, (ii) of a sufficient demand or aggregate 'willingness to pay' for such services by the potential beneficiaries, (iii) to delineate and enforce property rights surrounding land use and ES, and (iv) of investments in social capital to foster collective action and cohesion between the providers and beneficiaries of ES (Pagiola et al., 2004; Rosa et al., 2004; Tomich et al., 2004; Wunder, 2005; van Noordwijk et al., 2005). Note that property rights regimes in natural resource management comprise a structure of rights to resources, rules under which those rights are exercised, and duties bound by both those who possess the right(s) and those who do not. As Bromley (1991, pp. 2) puts it, "[p]roperty is not an object but rather is a social relation that defines the property holder with respect to something of value... against all others." In this context, Costa Rica is one of the few examples where an elaborate, nationwide PES program is in place. Under this program only farmers with property rights to land can be paid for the environmental conservation they provide (Pagiola, 2002).

A recent illustrative example of the potential effectiveness and flexibility of P(R)ES programs is that of RUPES approach: Rewarding Upland Poor for Environmental Services. The RUPES partnership of the International Fund for Agricultural Development (IFAD), the World Agroforestry Centre (ICRAF) and a partnership of local, national and international partners.[1] RUPES aims to conserve environmental services at the global and local levels while at the same time support the livelihoods of the upland poor in Asia. So far, the main focus has been on Nepal, Philippines and Indonesia and the environmental services mostly include water flow and quality from watersheds, biodiversity protection and carbon sequestration. Regarding, the demonstration, capture and sharing of benefits, the preliminary learning stock from the ongoing various RUPES experiences, include the following (van Noordwijk et al., 2005):

(i) *Demonstrating values* through scientific evidence of the link between ES and benefits under various land practices: In one RUPES site, Lake Singkarak in Sumatra, Indonesia, a major conclusion from an hydrological assessment conducted by ICRAF has been that reforesting the watershed may not significantly change the water inflows into the lake, which is originally what the local hydro-electrical company (the local ES buyer) is most interested in. This has implied questioning the (*a priori*) rationale for rewarding reforestation initiatives. Instead, the appraisal has identified water quality in the lake and the multiple sources of pollution as more important issues that would benefit both the hydro-electrical company and the local communities within the watershed.

(ii) *Capturing benefits* by identifying the potential beneficiaries/buyers: The RUPES experience is showing that localized buyers are more easily identifiable for effective partnership than regional or even global buyers. This implies that besides water conservation services, which may be more tangible for potential local buyers such as hydropower companies, biodiversity conservation and/or carbon sequestration pose more challenges given the difficulty to quantify the values that may justify a payment/reward for their sustained provision. In addition, identifying the providers of such services is also more elusive, due to their global public nature.

(iii) *Sharing* benefits by creating an enabling environment for sustaining the ES agreements by identifying potential institutional constraints: In this case, RUPES acknowledges that both property rights, especially when *de facto* (non *de jure*) rights for resource control are prevalent, and social capital, which helps to foster collective action at the local community level, are the two foremost important enabling factors.

4.2. Direct compensation payments (DCP)

A variant of P(R)ES, is the approach based on direct compensation payments (DCP) for 'takings' of landowners' private land out of production and into conservation (Swart, 2003). While theoretically sound in principle, there are important issues to be considered. First, similar to other incentive mechanisms, the identification of the level of the efficient compensation payments to landowners requires the demonstration of an objective measure of its conservation value on both biological and economic grounds. Second, the change in decentralised behaviour needs to be sustained into the future which requires longer term political commitment. Third, there is a more subtle but more problematic issue at play. It involves the existence of asymmetric information between landowners and the compensating government agency. This informational problem can create perverse incentives that reduce the effectiveness of the compensation mechanism (Innes et al., 1998). For instance, if landowners expect a compensation payment which is lower than the present value of the benefit stream arising from developing the land holding, they have a motive to develop their holdings in the 'first period', i.e., before being compensated in a subsequent period. This would have potentially negative effects on biodiversity conservation. But from the landowners' viewpoint, it reduces the risk of losing the land through the government's 'takings' for conservation purposes.

Furthermore, even when the exact compensation is foreseen by landowners, that is, the compensation coincides with the foregone expected agricultural revenues, they may still have the incentive to develop their land further by over-investing, e.g. added intensification, before any compensation is offered. This is because the market value of their

[1] Some of the insights reflected here come from personal communication with Meine van Noordwijk, Tom Tomich and ICRAF personnel involved in RUPES program in Sumatra, Indonesia.

property may increase due to such investments and such market value is what the government is guaranteeing as full compensation.

Thus, landowners' strategic behaviour exploiting existing information asymmetries, can seriously undermine the effectiveness of DCP mechanisms. One solution would be to offer relatively high (more than full) compensation to owners of underdeveloped (and hence biodiversity richer) land property compared to over-developed property owners as this counters the perverse intensification strategy through overinvestment. However, as Innes et al. (1998) note this strategy could significantly increase the public implementation bill, thus undermining its attractiveness from a cost-efficiency perspective.

4.3. Transferable development rights (TDR)

An interesting and cost-effective way to resolve the perverse incentives arising from DCPs is the use of transferable (land) development rights (TDR). TDR extend the longstanding 'agro-ecological zoning' schemes, which aim to direct development to areas of high productivity potential and to restrict agricultural land use in ecologically significant and sensitive areas. However, such zoning programs do not allow for any substitutability between plots in meeting overall conservation goals. By providing a market-like alternative to the DCPs, flexibility in achieving conservation goals can be introduced. In this vein, the main advantage of TDR is that it can, in principle, encourage conservation on lands with low agricultural opportunity costs, while providing appropriate incentives to the affected landholders (Panayotou, 1994; Chomitz, 1999).

In contrast to DCP, each landowner is issued tradable development permits by the government agency at an initial period. Subsequently, landowners hold the right to either develop/intensify their land holding. However, to develop that fraction of land a landowner needs to either use of one of the development permits (s)he holds or buy it from other landowners, who upon selling it can no longer develop their land fraction and instead must give it up for conservation. In this case, the government can share the cost of the 'takings', i.e., compulsory government land acquisition, with the landowners themselves.

Two main types of TDR programs exist at the landscape level: the single and dual zoning program. The former is similar to permit systems such as those used in transferable fishing quotas or pollution control. After the initial allocation of quotas, anyone within the program area may buy or sell the permits. An application of this type of such TDRs program has been used to control soil degradation through erosion in the Lake Tahoe Basin (Johnston and Madison, 1997). The dual zone system instead explicitly designates both (permit) sending and receiving areas. This allows, for example, for new land use restrictions to be imposed on the sending zone that is more ecologically sensitive, upon obtaining additional information about its higher conservation value and assigning TDRs to compensate for such additional restrictions. Usually, tight restrictions are also imposed on the receiving zone so as to increase the demand for TDRs (Chomitz, 1999).

One of the forerunners of the TDR mechanism is Brazil. While some initiatives have been proposed, the implementation is still under discussion. The basic idea is to give the opportunity for Brazilian agricultural land owners not complying with the National Forest Code (Law number 4771 approved on 15th September 1965) to buy forest reserves in other areas, normally in close proximity to his/her property. However, a fully operational market for forest reserves is still to be implemented. Two examples are the National Provisionary Measure (Medida Provisória, Number 21666-67, approved on 24th August 2001), which amends the Forest Code and in the State of Sao Paulo (State Decree number 50889, approved on 16th June 2006).

For agrobiodiversity conservation, the effectiveness of the TDR scheme relies on whether the objective is to conserve certain habitats within the landscape due to having unique biodiversity characteristics, or if larger tracks of contiguous habitats are necessary for off-farm biodiversity. When the landscape is highly homogeneous, and the goal is to conserve a specified 'amount' of habitat within the landscape, regardless of its configuration, a single zone system may be more appropriate.

While there is a theoretically attractive incentive mechanism, few rural TDR programs exist. This is possibly due to the political barriers. In fact, as with any tradable permit scheme, the initial allocation of permits is a sensitive issue that may have large distributional consequences (Chomitz, 1999). In addition, transaction costs also need to be taken into account as setting up TDRs may involve substantial administrative and legal (monitoring and enforcement) costs.

4.4. Auction contracts for conservation (ACCs)

One other way to achieve a desired level of supply of agrobiodiversity conservation at the landscape level by private landowners is by applying a competitive bidding or auction mechanism. An auction is a quasi-market institution with an interesting feature, i.e.: It has a 'cost revealing' advantage compared to P(R)ES and DCP and can, in principle, be incorporated into a TDR system. In fact, the cost-revelation feature provides an edge to generate important cost savings to governments. This is especially so when significant information asymmetry between farmers and conservation agencies exist regarding (i) the real opportunity cost of conservation and (ii) the ecological significance of the natural assets existing in farmlands. While the former is often better known by farmers themselves, the latter is normally better known by environmental experts (Latacz-Lohmann and van der Hamsvoort, 1997). As discussed above, such information asymmetries become a potent reason for missing agrobiodiversity conservation markets. The idea is to use auctions to reveal the hidden information needed to recreate voluntary conservation contracts between landholders and the government.

In essence, landholders submit bids to win conservation contracts from the government. But, while the latter prefers low bids, landowners need to submit bids that at least cover the opportunity cost of carrying out conservation activities in their farms. The problem is that information of such opportunity costs are often better known by farmers than by the government and they are also likely to be farmer-specific.

Stoneham et al. (2007) provides a recent small scale pilot case-study of an auctioning system for biodiversity conservation contracts in Victoria, Australia, known as BushTender. The ACC involved 98 farmers from which 75% obtained government contracts to conserve remnant vegetation in their farms, after all farmers submitted sealed bids associated with their nominated conservation action plans. The selection of the farmers who won the contract was based on ranking the relative cost-effectiveness of each proposed contract. This involved weighting each private bid against the associated potential ecological impacts at the landscape level. Given a public budget of US$ 400,000, contracts with bids that averaged about US$ 4600, were allocated and specified in management agreements over a 3-year period. In total the contracts covered 3160 ha of habitat on private land.

Stoneham et al. (2007) have estimated that the BushTender mechanism has provided 75% more biodiversity conservation compared to a fixed-price payment scheme (or DCP). In addition, they contend that given the relatively few enforcement costs in their pilot study, this ACC has interesting cost-effective properties. The pilot case-study shows that it is possible to recreate the supply side of a market for agrobiodiversity conservation.

All P(R)ES, DCP, TDPs and ACCs share an important characteristic for successful market creation, and that depends on the provision of good and accurate information at the demonstration, capture and sharing stages. If it is not possible, or very costly, to convey clear and credible information about the nature of the services derived from biodiversity, then the perception by the demanders as to how much they are willing to pay for such services would be distorted. Moreover, it would be naïve to champion market creation for biodiversity conservation if other supporting institutions are lacking. Furthermore, in general, if markets for agrobiodiversity are recreated without proper institutional and regulatory back-up, then the social costs of such policies may well outweigh the benefits from conservation (Barrett and Lybbert, 2000).

In a second-best world where information is elusive, most policy initiatives pragmatically focus on ensuring that institutions are developed so as to keep future options open (Tomich et al., 2004). In fact, most conservation policies are aiming at developing flexible and open institutions that can mitigate the negative effects of intensification in agroecosystems, without foreclosing future (de)intensification options.

5. Conclusions

In this paper we have discussed the institutional issues involved in the creation of market-like mechanisms for agrobiodiversity conservation. Since the causes of farmers' decisions to 'disinvest' in agrobiodiversity as an asset lie in the incentives offered by current markets and other institutions, the solution lies in corrective institutional design. We interpret changes in agrobiodiversity as the product of explicit or implicit decentralised farm-level decisions whose effects include both farm and landscape level changes in a range of ecosystem services. The solution is to develop mechanisms that provide a different set of incentives.

We close with two observations. The first is that the importance of interdisciplinary research on biodiversity in both traditional and modern agro-ecosystems is recognized as a prerequisite for the development of more effective agrobiodiversity conservation regimes (Jackson et al., 2005; Perrings et al., 2006). In order to evaluate the social consequences of agricultural practices that cause the local extirpation of species, the fragmentation of habitats or the change in the relative abundance of species, we need to better understand three interconnected aspects: (i) the role of biodiversity in agroecosystem functioning and processes, (ii) the way that changes in functioning and processes affect ecosystem services, and (iii) the impact of changes in services on the production of goods and services that are directly valued by people on- and off-agricultural landscapes.

The sustainability of agricultural landscapes may involve a continuum of existing farm management systems from modern, intensive, mechanized, high-input, high-output systems at one end to traditional, extensive, labor-intensive, low-input, low-output systems at the other. Since the unit of analysis is the landscape, it may even be possible that an effective strategy is to have an extreme combination of highly intensive agriculture combined with low intensively managed areas (Green et al., 2005; Dorrough et al., 2007). Since the effects of such strategy can be different in landscapes that still contain wilderness areas, such in tropical forest margins, and in already ecologically impoverished agroecosystems, further collaborative research between ecologists and economists is identified as a high priority. In addition, often the alternative to intensification frequently involves encroachment on ever more marginal land and the destruction and fragmentation of ever more scarce habitat. But intensification that ignores the costs of a change in the mix of species in the system may be even more harmful. The point is, though, that this is an empirical question and that the research needed to identify the optimal mosaic has yet to be done. Alongside this point of view is the ongoing effort to advocate in favour of a biodiversity-based agriculture that can be managed in a way that can still produce high yields (Jackson et al., 2007).

The second observation is that in a sector where the impact on biodiversity is in the hands of billions of independent land-holders, management of agrobiodiversity by direct centralized control is not an option. What is important is that independent decision-makers take into account the true social costs and benefits of their actions. For example, farmers who maintain production of drought or disease resistant crops or livestock confer social benefits (in terms of averting expenditures on famine relief) that are seldom reflected in the prices they receive. Whether this implies taxation of the high risk components or subsidy of the low risk components depends on local circumstances and the international trading regime. In other words, the effectiveness of alternative mechanisms for changing farmers' decisions is also an empirical question. While it may be possible to identify the social opportunity cost of alternative farm management strategies, the best method for inducing socially optimal behavior depends on understanding not just the responsiveness of farmers and consumers, i.e., the relevant elasticities but also the role of the social, cultural and institutional environment.

As in the European Union, in many parts of the world, perverse subsidies are being morphed into direct compensation payments to providers of the non-marketed agrobiodiversity services or used to convert the overhead costs of setting up direct (e.g. DCP) or/and indirect incentive schemes, e.g., P(R)ES, TDR and ACC. While there is considerable advantage in removing the perverse incentive effects of historic subsidies, few of the current agricultural reforms are based on a serious valuation of the social opportunity cost of agrobiodiversity loss, and fewer still involve an appraisal of the allocative effects of the new payment schemes. Sensible design of market-like mechanisms for agrobiodiversity conservation requires both.

Acknowledgements

We would like to thank Kamal Bawa, George Brown, Louise Jackson, Danilo Igliori, Andreas Kontoleon, Esti Orruño, Per Stromber, Tom Tomich and Meine van Noordwijk for useful comments and suggestions to previous drafts of this paper. We also extend out thanks to two anonymous referees.

References

Altieri, M.A., 1999. The ecological role of biodiversity in agroecosystems. Agric. Ecosyst. Environ. 74, 19–31.

Bacon, C., 2005. Confronting the coffee crisis: can fair trade, organic and specialty coffees reduce small-scale farmer vulnerability in Northern Nicaragua? World Dev. 33, 497–511.

Barbier, E.B., 1989. The Economic Value of Ecosystems. 1-Tropical Wetlands. Gatekeeper No LEEC 89-02. London Environmental Economics Centre (LEEC), London.

Barbier, E.B., 2000. The economic linkages between rural poverty and land degradation: some evidence from Africa. Agric. Ecosyst. Environ. 82, 355–370.

Barrett, C.B., Lybbert, T.J., 2000. Ecol. Econ. Is bioprospecting a viable strategy for conserving tropical ecosystems? 34, 293–300.

Baumgärtner, S., submitted. The insurance value of biodiversity in the provision of ecosystem services. Nat. Resour. Mod.

Bélanger, L., Grenier, M., 2002. Fragmentation increased along a gradient from traditional dairy agriculture to more intensive cash crop agriculture. Landsc. Ecol. 17, 495–507.

Benton, T.G., Bryant, D.M., Cole, L., Crick, H.Q.P., 2002. Linking agricultural practice to insect and bird populations: a historical study over three decades. J. Appl. Ecol. 39, 673–687.

Birol, E., Smale, M., Gyovai, Á., 2006. Using a choice experiment to estimate farmers' valuation of agricultural biodiversity on Hungarian small farms. Environ. Resour. Econ. 34 (4), 439–469.

Carpenter, S., Walker, B., Anderies, J., Abel, N., 2001. From metaphor to measurement: resilience of what to what? Ecosystems 4, 765–781.

Chomitz, K.E., 1999. Transferable development rights and forest protection: an exploratory analysis. Paper prepared for the Workshop on Market-Based Instruments for Environmental Protection. July, 1999. John F. Kennedy School of Government, Harvard University.

Cobb, D., Dolman, P., O'Riordan, T., 1999. Interpretations of sustainable agriculture in the UK. Prog. Hum. Geog. 23, 209–235.

Conway, G.R., 1993. Sustainable agriculture: the trade-offs with productivity, stability and equitability. In: Barbier, E.B. (Ed.), Economics and Ecology New Frontiers and Sustainable Development. Chapman and Hall, London, pp. 45–65.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.

Crocker, T.D., Tschirhart, J., 1992. Ecosystems, externalities, and economics. Environ. Resour. Econ. 2, 551–567.

Dasgupta, P., 2000. Entry on valuing biodiversity. In: Levin, S., Daily, G.C., Lubchenco, J., Tilman, D. (Eds.), Encyclopedia of Biodiversity. Academic Press.

Di Falco, S., Perrings, C., 2003. Crop genetic diversity, productivity and stability of agroecosystems. A theoretical and empirical investigation. Scot J. Polit. Econ. 50, 207–216.

Di Falco, S., Perrings, C., 2005. Crop biodiversity, risk management and the implications of agricultural assistance. Ecol. Econ. 55, 459–466.

Dorrough, J., Moll, J., Crosthwaite, J., 2007. Can intensification of temperate Australian livestock production systems save land for native biodiversity? Agric. Ecosyst. Environ. 121, 222–232.

Folke, C., Holling, C.S., Perrings, C., 1996. Biological diversity, ecosystems and the human scale. Ecol. Appl. 6, 1018–1024.

Green, R.E., Cornell, S.J., Scharlemann, P.W., Balmford, A., 2005. Farming and the fate of wild nature. Science 307, 550–555.

Hodge, I., 2000. Agri-environmental relationships and the choice of policy mechanism. World. Econ. 23, 257–273.

Holling, C.S., 1988. Temperate forest insect outbreaks, tropical deforestation and migratory birds. Memoirs Entomol. Soc. Canada 146, 21–32.

Holling, C.S., 1996. Engineering resilience versus ecological resilience. In: Schulze, P. (Ed.), Engineering Within Ecological Constraints. National Academy, Washington (DC), pp. 31–44.

Innes, R., Polasky, S., Tschirhart, J., 1998. Takings, compensation and endangered species protection on private lands. J. Econ. Perspect. 12, 35–52.

Jackson, L.E., Bawa, K., Pascual, U., Perrings, C., 2005. AgroBIODIVERSITY: a new science agenda for biodiversity in support of sustainable agroecosystems. DIVERSITAS report. N. 4. 40 pp.

Jackson, L.E., Pascual, U., Hodking, T., 2007. Utilizing and conserving agrobiodiversity in agricultural landscapes. Agric. Ecosyst. Environ. 121, 196–210.

Johnston, R., Madison, M., 1997. From landmarks to landscapes: a review of current practices in the transfer of development rights. J. Am. Plan. Assoc. 63, 365–378.

Kremen, C., Williams, N.M., Thorp, R., 2002. Crop pollination from native bees at risk from agricultural intensification. Proc. Natl. Acad. Sci. USA 99, 16812–16816.

Lambin, E.F., Turner, I.I.B.L., Geist, H.J., Agbola, S., Angelsen, A., Bruce, J.W., Coomes, O., Dirzo, R., Fischer, G., Folke, C., George, P.S., Homewood, K., Imbernon, J., Leemans, R., Li, X., Moran, E.F., Mortimore, M., Ramakrishnan, P.S., Richards, J.F., Skånes, H., Steffen, W., Stone, G.D., Svedin, U., Veldkamp, T., Vogel, C., Xu, J., 2001. The causes of land-use and land-cover change: moving beyond the myths. Global Environ. Chang. 11, 261–269.

Latacz-Lohmann, U., van der Hamsvoort, C., 1997. Auctioning conservation contracts: a theoretical analysis and an application. Am. J. Agric. Econ. 79, 407–418.

Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J.P., Hector, A., Hooper, D.U., Huston, M.A., Raffaeli, D., Schmid, B., Tilman, D., Wardle, D.A., 2002. Biodiversity and ecosystem functioning: current knowledge and future challenges. Science 294, 804–808.

MEA (Millennium Ecosystem Assessment), 2005. Ecosystems and Human Well-being: Biodiversity Synthesis. World Resources Institute, Washington, DC.

Mooney, H.A., Mack, R.N., McNeely, J.A., Neville, L.E., Schei, P.J., Waage, J.K. (Eds.), 2005. Invasive Alien Species: A New Synthesis. Island Press, Washington, DC.

Norton, B.G., 1986. On the inherent danger of undervaluing species. In: Norton, B.G. (Ed.), The Preservation of Species. Princeton University Press, Princeton.

OECD (Organisation for Economic Co-operation Development), 1993. What future for Our Countryside? A Rural Development Policy. OECD, Paris.

OECD (Organisation for Economic Co-operation and Development), 2005. Handbook of Market Creation for Biodiversity: Issues in Implementation. Organisation for Economic Co-operation and Development. OECD, Paris.

Opschoor, J.H., 1999. Making the benefits of biodiversity conservation visible and real: institutional aspects in a biodiversity research programme. Environ. Dev. Econ. 4, 204–214.

Östman, O., Ekbom, B., Bengtsson, J., 2003. Yield increase attributable to aphid predation by ground-living polyphagous natural enemies in spring barley in Sweden. Ecol. Econ. 45, 149–158.

Pagiola, S., 2002. Paying for water services in Central America: learning from Costa Rica. In: Pagiola, S., Bishop, J., Landell-Mills, N. (Eds.), Selling Forest Environmental Services: Market-based Mechanisms for Conservation and Development. Earthscan, London.

Pagiola, S., Agostini, P. Gobbi, G., de Haan, G., Ibrahim, M. Murgueitio, E., Ramirez, E., Rosales, M., Ruiz, J.P., 2004. Paying for biodiversity conservation services in agricultural landscapes. Environment department paper no. 96. World Bank, Washington DC.

Panayotou, T., 1994. Conservation of biodiversity and economic development: the concept of transferable development rights. Environ. Resour. Econ. 4, 91–110.

Pearce, D.W., 1999. Economics and Environment: Essays on Ecological Economics and Sustainable Development. Edward Elgar, Cheltenham.

Perrings, C., 1998. Resilience in the dynamics of economy-environment systems. Environ. Resour. Econ. 11, 503–520.

Perrings, C., 2001. The economics of biodiversity loss and agricultural development in low income countries. In: Lee, D.R., Barrett, C.B. (Eds.), Tradeoffs or Synergies? Agricultural Intensification, Economic Development and the Environment. CAB International, Wallingford, pp. 57–72.

Perrings, C., 2005. Mitigation and adaptation strategies for the control of biological invasions. Ecol. Econ. 52, 315–325.

Perrings, C., Gadgil, M., 2003. Conserving biodiversity: reconciling local and global public benefits. In: Kaul, I., Conceicao, P., le Goulven, K., Mendoza, R.L. (Eds.), Providing Global Public Goods: Managing Globalization. OUP, Oxford, pp. 532–555.

Perrings, C., Jackson, L., Bawa, K., Brussaard, L., Brush, S., Gavin, T., Papa, R., Pascual, U., de Ruiter, P., 2006. Biodiversity in agricultural landscapes: saving natural capital without losing interest. Conserv. Biol. 20, 263–264.

Peterson, G., Allen, C.R., Holling, C.S., 1998. Ecological resilience, biodiversity, and scale. Ecosystems 1, 6–18.

Ricketts, T.H., Daily, G.C., Ehrlich, P.R., Michener, C.D., 2004. Economic value of tropical forest to coffee production. Proc. Natl. Acad. Sci. USA 101, 12579–12582.

Rodríguez, L.C., Pascual, U., Niemeyer, H.M., 2006. Peasant communities' cultural domain and the local use-value of plant resources: the case of Opuntia scrublands in Ayacucho, Peru. Ecol. Econ. 57, 30–44.

Rosa, H., Kandel, S., Dimas, L., 2004. Compensation for environmental services and rural communities: lessons from the Americas. Int. Forest. Rev. 6, 187–194.

Scherr, S.J., McNeely, J.A., in press. Biodiversity conservation and agricultural sustainability towards a new paradigm of 'ecoagriculture' landscapes. Philos. T. Roy. Soc.

Schläpfer, F., Tucker, M., Seidl, I., 2002. Returns from hay cultivation in fertilized low diversity and non-fertilized high diversity grassland. Environ. Resour. Econ. 21, 89–100.

Smale, M., Hartell, J., Heisey, P.W., Senauer, B., 1998. The contribution of genetic resources and diversity to wheat production in the Punjab of Pakistan. Am. J. Agr. Econ. 80, 482–493.

Smale, M., Bellon, R.M., Aguirre Gomez, J.A., 2001. Maize diversity, variety attributes, and farmers' choices in Southeastern Guanajuato, Mexico. Econ. Dev. Cult. Change 50, 201–225.

Stoneham, G., Chaudhri, V., Strappazzon, L., Ha, A., 2007. Auctioning biodiversity conservation contracts. In: Kontoleon, A., Pascual, U., Swanson, T. (Eds.), Biodiversity Economics: Principles, Methods and Applications. Cambridge University Press, Cambridge.

Swanson, T. (Ed.), 1998. The Economics and Ecology of Biodiversity Decline: The Forces Driving Global Change. Cambridge University Press, Cambridge.

Swart, J.A.A., 2003. Will direct payments help biodiversity? Science 299, 1981.

Swift, M.J., Anderson, J.M., 1993. Biodiversity and ecosystem function in agroecosystems. In: Schultze, E., Mooney, H.A. (Eds.), Biodiversity and Ecosystem Function. Springer, New York, pp. 57–83.

Symondson, W.O.C., Sunderland, K.D., Greenstone, M.H., 2002. Can generalist predators be effective biocontrol agents? Annu. Rev. Entomol. 47, 561–594.

ten Kate, K., Laird, S.A., 1999. The Commercial Use of Biodiversity—Access to Genetic Resources and Benefit-Sharing. Earthscan, London.

Tilman, D., Wedin, D., Knops, J., 1996. Productivity and sustainability influenced by biodiversity in grasslands ecosystems. Nature 379, 718–720.

Tilman, D., Cassman, K.G., Matson, P.A., Naylor, R., Polasky, S., 2002. Agricultural sustainability and intensive production practices. Nature 418, 671–677.

Tomich, T.P., van Noordwijk, M., Budidarsono, S., Gillison, A., Kusumanto, T., Murdiyarso, D., Stolle, F., Fagi, A.M., 2001. Agricultural intensification, deforestation, and the environment: assessing tradeoffs in Sumatra, Indonesia. In: Lee, D.R., Barrett, C.B. (Eds.), Tradeoffs or Synergies? Agricultural Intensification, Economic Development and The Environment. CAB-International, Wallingford, UK, pp. 221–244.

Tomich, T.P., Thomas, D.E., van Noordwijk, M., 2004. Environmental services and land use change in Southeast Asia: from recognition to regulation or reward? Agric. Ecosyst. Environ. 104, 229–244.

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005. Landscape perspectives on agricultural intensification and biodiversity ecosystem service management. Ecol. Lett. 8, 857–874.

Turner, R.K., Pearce, D.W., 1993. Sustainable economic development: economic and ethical principles. In: Barbier, E.B. (Ed.), Economics and Ecology: New Frontiers and Sustainable Development. Chapman & Hall, London.

van Noordwijk, M., Kuncoro, S., Martin, E., Joshi, L., Saipothong, P., Areskoug, V., O'Connor, T., 2005. Donkeys, carrots, sticks and roads to a market for environmental services: rapid agrobiodiversity appraisal for the PES–ICDP continuum. In: Paper presented at the DIVERSTIAS First Open Science Conference, Oaxaca, November.

Vandermeer, J., Perfecto, I., 1995. Breakfast of Biodiversity: The Truth about Rainforest Destruction. Food First Books, Oakland, CA, USA.

Widawsky, D., Rozelle, S.D., 1998. Varietal diversity and yield variability in Chinese rice production. In: Smale, M. (Ed.), Farmers, Gene Banks, and Crop Breeding. Economic Analyses of Diversity in Wheat, Maize, and Rice. Kluwer, Boston, pp. 159–172.

Wunder, S., 2005. Payments for Environmental Services: Some Nuts and Bolts. CIFOR Occasional paper. Centre for International Forestry Research. Bogor Barat, Indonesia.