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CHAPTER 3

OPPORTUNITIES AND GAPS IN PES IMPLEMENTATION AND KEY AREAS FOR FURTHER INVESTIGATION

Daniela Ottaviani
Food and Agriculture Organization of the United Nations, Rome, Italy

CONTENTS

ABSTRACT

Payment for Ecosystem Services (PES) is often considered a simple market tool conceived to reflect the value of positive externalities related to the provision of ecosystem services. Having a clear economic structure the performance of PES programmes is often evaluated by economic criteria, such as additionality, economic efficiency, conditionality, leakage and permanence of benefits. However, emphasis on the economic structure of PES schemes has often hidden the ecological and social dimensions that are linked the fundamental purpose of PES.

The understanding of positive relationships (synergies) and negative interactions (trade-offs) occurring amongst the multiple ecosystem functions is key to designing PES schemes that are more efficient in the delivering ecosystem services for society. Ecosystem services have a social value because they are natural capital belonging to the whole of society. Reflecting the value of ecosystem services is likely to involve different stakeholders at the local, regional and global scales, which can lead to social debate and conflicting views. PES should, thus, reflect societal preferences, which are not just the sum of individual preferences; reflecting societal consensus should be completely driven by a participatory approach. Although PES was not originally conceived as a tool for poverty alleviation, some elements of PES design can increase the potential for this. As such the possibility of implementing community PES programmes seems a major opportunity for the new generation of PES schemes in which all community members could receive some benefits.

Integrating economic, ecological and social criteria in PES design and implementation will certainly increase its complexity, but this integration could lead PES to support sustainability by promoting economic resilience, environmental integrity and social development.

PES: BEYOND A SIMPLE MARKET TOOL

PES schemes have a clear economic structure constituted by voluntary contractual agreements that define economic transactions between a buyer and a seller for the provision of ecosystem services. Due to this economic structure, PES is often thought of as a simple economic incentive that is operated and regulated by economic principles and market rules. This emphasis on the economic dimension has often hidden the ecological and social dimensions of PES schemes though. However, it is the ecological and social dimensions that are expressed in the fundamental purpose of PES: to preserve the functioning of ecosystem services for the well-being of society. Integrating the multiple dimensions of PES requires consideration of different criteria, inputs, processes and a different level of dialogue.

Integrating the multiple dimensions of PES requires consideration of different criteria, inputs, processes and levels of dialogue

Costanza and Folke (1997) point out that, in the economic dimension, PES schemes require only a low level of scientific input and discussion amongst stakeholders to achieve economic efficiency. In fact, the definition of a market value for the ecosystem services can be assessed according to individual preferences, which do not require agreement or social consensus. Moreover, the market value can simply reflect individually perceived opportunity costs, which do not necessarily require an understanding of the biophysical linkages in the ecosystem functioning.

On the contrary, in the ecological dimension, PES schemes need a high level of scientific input if they aim to conserve or restore ecosystem services (Kremen, 2005). The highly specialised nature of this input will require a certain degree of filtering and simplification through the use of modelling and scenarios to be able to be shared and discussed amongst stakeholders coming from different backgrounds.

In the social dimension, PES schemes need a medium level of scientific input that constitutes the background information needed to set up an active participatory social debate (Costanza and Folke, 1997). However, intensive dialogue will be required to define equity and justice criteria for the distribution of resources and property rights both with the current and future generations.

By considering the multidimensional nature of PES in the first section of this chapter, it is seen that the main economic attributes of PES are often detrimental to the expression of the ecological and social dimensions. In particular, the assumptions upon which PES schemes are based that define current market-based criteria for PES performance are critically reviewed. In the next section, the importance of ecosystem processes and functions for the provision of ecosystem services will be revisited. The importance of market restrictions for natural resources and the need to assess and model the provision of multiple ecosystem services is highlighted. In the third section of this chapter, the value of ecosystem services for society and the different perspectives of stakeholders at the local, regional and global scales will be looked into. Finally, the potential of PES for poverty alleviation and important factors that might be crucial in the next generation of community-based PES schemes are discussed.

UNDERLYING ECONOMIC ASSUMPTIONS OF PES

The economic rationale of PES as a market tool that provides positive incentives to ensure the delivery of ecosystem services to society mainly reflects different principles of a neoclassical economic framework. These principles include the utilitarian anthropocentric principle, market essentialism and consumer choice theory, and the optimistic predictions of the Coase Theorem[1] (Coase, 1960).

1 The Coase Theorem states that when private property rights are clearly defined by enforceable contracts, then the supplier and buyer of an externality can, through voluntary exchange, potentially reach an agreement that maximises social welfare.

All ecosystem functions are considered ecosystem services only if there are people that can benefit from their delivery. This utilitarian principle also brings the idea that ecosystem services have an economic value only if people consider them valuable and are willing to pay for them. According to market essentialism, markets, surrogate markets and simulated markets are the ideal institutions for the efficient allocation of resources that will adequately quantify the monetary value of public goods and signal their scarcity through price fluctuations.

Within the market mechanism, individuals are expected to behave according to the consumer choice theory under which: (a) individuals are mainly self-interested and act as rational actors to maximise the utility (i.e. satisfaction of their preferences); (b) they can make rational choices because their decisions are based on complete information and reliable forecasts on the likelihood of possible outcomes; (c) they have a single, stable, invariant set of preferences, which are internally consistent and structured; and (d) they have preferences whose strength can be measured by their willingness to pay (WTP) for a degree of satisfaction or a willingness to accept compensation (WTA) for benefits forgone (Chee, 2004). The optimistic attitude towards PES schemes is also based on the Coase Theorem, which states that when private property rights are clearly defined by enforceable contracts, the supplier and buyer of an externality can then, through voluntary exchange, potentially reach an agreement that maximises social welfare. However, Coase (1960) himself argued that this outcome will only occur in the absence of wealth effects and transaction costs (Chee, 2004).

Neoclassical economic frameworks do not reflect complexity of socio-economic contexts or the drivers underlying individual choices

It is clear that these economic assumptions often do not reflect the complexity of socio-economic contexts, nor the diversity of drivers underlying individual choices. This gap can be counterbalanced though when the ecological and social dimensions are fully tackled by PES schemes. In this case, the disruption of ecosystem services will not be expected to be signalled only by price fluctuations; instead, there will be a robust scientific background to provide scenarios for policy and decision making. At the same time, the participatory nature of the social dimension of PES will enhance stakeholder dialogue and allow societal preferences to be born and negotiated, eventually resulting in community consensus and collective action.

CURRENT MARKET-BASED CRITERIA OF PES PERFORMANCE

Five criteria are generally used to evaluate economic performances of PES programmes including: (a) additionality, (b) economic efficiency, (c) conditionality, (d) leakage, and (e) permanence of benefits.

a. **Additionality** requires that any change/improvement/adoption of a different practice should be additional to the scenario that would have occurred in the absence of the PES project. To lack additionality means that PES programmes are paying for something that would have been adopted anyway, which results in poor financial efficiency. Therefore, projects must demonstrate actions over and above 'business as usual'. This additionality criterion also implies that, in the valuation of multiple ecosystem services, 'double counting' should be avoided as creating uncertainty and poor reliability of the valuation. Because ecosystems always provide multiple ecosystem services the most common approach is to try to value single ecosystem services independently and then add all the obtained values together to obtain the total monetary value of ecosystem services in the ecosystem. Fu *et al.* (2010) suggest different measures to reduce the probability of double-counting though. However, additionality seems an inappropriate standard when dealing with ecosystems, which are constituted by multiple non-linear interactions amongst ecosystem services.
b. **Economic efficiency** (i.e. the optimal allocation of resources) in PES involves maximising the differences between benefits and costs, where benefits are those obtained from the provision of ecosystem services and costs include opportunity costs of individual land properties, information and transaction costs. In particular, economic efficiency is often challenged by the difficulty of pinpointing the true opportunity cost. This is caused by the asymmetric information between the seller and the buyer of ecosystem services (i.e. while the seller of ecosystem services knows the opportunity costs given by the land he owns, the buyer does not know what the lowest price is at which the seller would be willing to accept the offer and engage in a PES scheme). PES programmes are considered economically inefficient when they pay more than the landowner's true opportunity cost.
c. **Conditionality** is a performance criterion upon which the definition of a PES contract is based. In fact, payment should be provided upon condition that the provision of the ecosystem service has been delivered. Due to the voluntary nature of PES agreements, it is assumed that any failure to meet the expected conditions (i.e. a lack of conditionality) will determine the end of the contractual agreement. In fact, according to the neoclassical economic framework, once a voluntary market agreement is established it reflects the highest goods for both the seller and buyer. In reality, conditionality is assumed but seldom verified, with serious consequences for the real evaluation of PES performance. When buyers are not direct users (e.g. in public-financed PES schemes), they do not have first-hand information and have little direct incentive to ensure that the programme is working efficiently. In addition, public-financed PES can be subject to a variety of political pressures.
d. **Leakage**, otherwise known as spillage, refers to the inadvertent displacement of activities damaging ecosystem services provision to areas outside the geographical zone of PES

intervention (Robertson and Wunder, 2005). Leakage may occur directly, for example, if landholders engaging with PES for the protection of forests on their lands shift deforestation activities to other areas. Leakage may also occur indirectly through market mechanisms, for example, land enrolment in PES for forest conservation may lead to increased prices of forest products, thereby encouraging extractive activities in other forest areas.

e. **Permanence of benefits** refers to the ability of a PES programme to achieve long-term improvements in ecosystem service provision, including beyond the period of the payments. Sometimes permanence is suggested as a criterion of PES performance. However, this criterion assumes a degree of stability of the *status quo* both on the ecological and socio-economic dimensions. From the ecological perspective, given that ecosystems are composed of multiple interacting ecosystem services, the stability of a certain ecosystem function cannot be expected over time due to unexpected disturbances and interventions that may occur via other interlinked ecosystem functions. Even if ecosystem complexity is excluded and a single ecosystem service is considered, the lack of ecological criteria in the PES design often hampers the ecological permanence of benefits. In absolute terms, ecological permanence in the delivery of ecosystem services cannot be expected given the high rate of catastrophes, the subtle ongoing changes currently affecting the planet and the many demands and ecological pressures placed on land. Socio-economic permanence in the delivery of ecosystem services is also not likely to occur unless PES initiatives are driven by strong individual and community motivational drivers. In fact, PES is considered an advanced market tool with a flexible structure, being a voluntary transaction based on a conditional agreement and, thus, able to adapt to political, economic and social changes. In principle, participants in PES programmes cannot be expected to continue to respect the contractual agreement once the payment is over. Numerous studies show that when people receive a monetary payment for doing something they would have done anyway, their motivation for doing it without payment diminishes; they also do it less well if they perceive the payment as inadequate and they may stop doing it altogether when payment ceases (Farley and Costanza, 2010).

MAINTAINING THE FUNCTIONING OF ECOSYSTEMS: THE ECOLOGICAL DIMENSION OF PES

The provision of services from an ecosystem depends on complex processes that must be recognised in the design of PES. The structure and composition of ecosystems will profoundly affect the provision of ecosystem services, such as water purification, carbon sequestration and pollination (see Viewpoint 3 “PES design: Inducing cooperation for landscape-scale ecosystem services management"). Understanding the characteristics of ecosystems that need to be preserved to maintain ecosystem functionality is an important first step towards incorporating these elements into PES design.

As described by Moss (2008), undisturbed natural ecosystems are characterised by a high level of resilience; they are self-maintaining, requiring no human management. Ecosystem resilience is linked to the preservation of ecosystem structure, size, connectivity and balance of chemical nutrients (Moss, 2008). Ecosystem structures include both physical (geomorphological features, tree debris, etc.) and biological components (food webs, keystone species, etc.); landscape connectivity, including both the spatial continuity between landscape elements (structural connectivity) and the response of individuals to landscape features (functional connectivity). Ecosystem size refers to occurrence of a sufficient area likely to include a sufficient variation in biological diversity, which will be able to cope with inevitable fluctuations in ecosystem conditions. A balanced amount of chemical nutrients is a property of a well-preserved ecosystem, which is commonly characterised by parsimony of available nutrients because most of them are tied up in the biological component and tightly recycled. Thus, an undisturbed natural ecosystem maintains its functionality because its size, structure and connectivity support a sufficient diversity of life forms that are able to efficiently recycle nutrients and ensure a balanced flow of matter and energy through the ecosystem. In summary, as suggested by Wallace (2007), the structure and composition of ecosystems highly influences ecosystem processes.

Ecosystem resilience is linked to the preservation of ecosystem structure, size, connectivity and the balance of nutrients

BIODIVERSITY: A KEY ATTRIBUTE OF ECOSYSTEMS NEEDED FOR THE PROVISION OF SERVICES

Biodiversity is a key attribute of ecological systems having a fundamental role in ecosystem functioning and, thus, in the provision of benefits to society or services (TEEB, 2009). Functioning is constituted by all the ecological processes controlling the fluxes of energy, nutrients and organic matter in the ecosystem. These fluxes are developed and regulated through the web of living organisms, which take in energy and substances, grow, reproduce, die and are decomposed back into the fluxes of organic matter, energy and nutrients throughout their life cycle. Thus, ecosystem functioning is based on primary production, decomposition and nutrient cycling. Every species is considered as having a unique ecological niche and consequently a higher number of species in a community should be able to more efficiently use resources, produce more biomass and show more resilience and adaptation to environmental changes than a community with a lower degree of biodiversity (Loreau *et al.*, 2001; Tilman, 1996).

Biodiversity loss occurs at different scales: locally as species richness decreases in biological communities and globally as the rate of species extinctions increases on the planet. The main direct drivers of biodiversity changes are habitat change, climate change, invasive species, over-exploitation, unbalanced nutrients and pollution (Sala *et al.*, 2000). The current increasing rate of biodiversity loss has raised some concerns that this might seriously affect ecosystem functioning and, thus, the ongoing provision of ecosystem services (TEEB, 2009).

From the early 1990s, many investigations have been carried out to identify and quantify the amount of biodiversity needed to ensure ecosystem functioning. The aim is to set up experimental conditions that enable a reduction of the number of species in an ecosystem and measure how this loss of diversity impacts key ecosystem processes. However, these findings are mainly constrained by three factors: (a) experiments are mostly carried out at a small scale and in over-simplified environments; (b) they mainly focus on only one component of biodiversity, which is easy to manipulate (e.g. terrestrial plants or algae); and (c) they often quantify the amount of biodiversity needed for the provision of a single ecosystem process in isolation, while few deal with multi-functionality of ecosystems (see Hector and Bagchi, 2007). The variability amongst the different experimental designs linked to the complexity of this field of investigation has made it very difficult to reach a consensus and a common framework. Recent meta-analyses (Balvanera *et al.*, 2006; Cardinale *et al.*, 2006; Quijas *et al.*, 2010; Schmid *et al.*, 2009, Worm *et al.*, 2006) of this extensive experimental work show the positive effect of biodiversity in the provision of most ecosystem services analysed. They also suggest that the relationship between species richness and many ecosystem functions, such as primary production and water and nutrient cycling, tend to be described by a saturating curve in both terrestrial and aquatic ecosystems (Cardinale *et al.*, 2006; Hector and Bagchi, 2007). The saturating effect is expected as the increased number of species in the community brings an increased overlap of ecological niches amongst species (Schmid *et al.*, 2009) and its main consequence is that the loss of some overlapping species may not decrease ecosystem functioning, but the loss of non-overlapping species will (Loreau *et al.*, 2002).

Fluxes of energy, nutrients and organic matter are developed and regulated in ecosystems through the web of living organisms

Biodiversity has positive effects at a community level and not at a population level; thus, populations are expected to fluctuate more with the increasing number of species in the community, while the species community is expected to record higher productivity and increased stability (Ives and Carpenter, 2007; Tilman, 1996). However, the stability (i.e. resilience) will vary with the type of disturbance taken into consideration. In particular, biodiversity is expected to have different effects on different trophic levels of an ecosystem. When the number of species belonging to one trophic level increases, this has a detrimental effect on the trophic levels below (top-down effect) and above (bottom-up effect).

On the other hand, increased species richness at a trophic level enhances its functionality and benefits symbiont species (Schmid *et al.*, 2009). Some of these predictions were evaluated in both brown (detritus–consumer) and green (plant–herbivore) food webs (Balvanera *et al.*, 2005, 2006; Cardinale *et al.*, 2006, 2009; Duffy *et al.*, 2007; Quijas *et al.*, 2010; Rey Benayas *et al.*, 2009; Schmid *et al.*, 2009; Srivasta and Vellend, 2005; Srivasta *et al.*, 2009). However, responses observed under experimental manipulations of a single trophic level may be more complex and difficult to predict in the real-life scenario of multi-trophic interactions occurring in ecosystems (Duffy *et al.*, 2007).

Some theoretical and experimental work is still needed to quantify in detail the relationship between biodiversity and ecosystem services, such as water quality, water quantity, pollination, regulation of pests and human diseases, carbon storage, climate regulation (Balvanera *et al.*, 2006; Kremen *et al.*, 2004).

Most of the existing experimental evidence focuses on species richness and it is clear that the number of species required to support multiple ecosystem services might be greater than considering a single ecosystem service (Hector and Bagchi, 2007). Moreover, ecosystem services might not be affected only by species richness, but also by species eveness (relative abundance of species) and species composition.

There are still substantial gaps in matching biodiversity components (populations, communities, functional groups, habitat types) to ecosystem functioning (Luck *et al.*, 2009). Thus, working under a precautionary principle fostering biodiversity conservation remains the major insurance facility for ecosystem service provision.

MARKETS FOR BIODIVERSITY: THE NEED FOR MARKET RESTRICTIONS

Around the world, different markets have been established for the trading of natural resources (Table 3). In these virtual markets, a development project that involves the depletion of natural stocks or an alteration of ecosystem processes can buy credits to offset the damage that the project activities will cause and compensate or mitigate these effects with the protection or restoration of an equivalency in a different place.

This trading system is expected to have a neutral effect (no net loss, no net gain) on the overall conservation status of biodiversity. It is clear that this is a simplistic way to approach the challenging target of biodiversity conservation. In particular, there is a fundamental mismatch between the economic principles that regulate common economic markets and the principles that can be applied to biodiversity trading.

Economic market rules require the use of a simple currency and the occurrence of minimal exchange restrictions to be able to build a free dynamic market. Moreover, for efficiency purposes,

Table 3
Examples of current markets of ecosystem goods and services

Resource	Market	Countries
Biodiversity	Biodiversity offsets are recognised in the legal framework of several countries	Australia, Brazil, Canada, Europe, New Zealand, USA
Fish stocks	Individual transferable fishing quota systems	Australia, New Zealand, Canada, Chile, Iceland, the Netherlands
Forests	Reducing Emissions from Deforestation and Forest Degradation (REDD)	Global
Vegetation	Bio-banking and net-gain initiatives	Australia
Water	Tradable permits for saline water discharges according to the Hunter River Salinity Trading Scheme	Australia
Wetlands	Wetland banking	USA

the review of implementation of market activities, if taking place, cannot be onerous. These three attributes of economic markets sharply contrasts with the characteristics of biodiversity markets. First, there is no simple currency able to capture the complexity of biodiversity. What is generally called 'biodiversity' indicates a hierarchical structure of diversity whose range extends from genes to ecosystems. When biodiversity is tackled at the ecosystem level, as in ecosystem services, then all levels of biodiversity are involved (genes, species, populations and communities). This implies that the biodiversity of the ecosystem will be the unique combination resulting from the interaction of the biodiversity recorded at the genetic, species and community levels. Moreover, ecosystem biodiversity is also an emergent property that arises from the combination and interaction of its single constituents. In nature, the possibility of finding an ecological unit which is like another is highly dependent on the appropriate consideration of the scale and the configuration in which the constituents of the unit are assembled. When the inner variability of the system is considered (genetic diversity), together with the variability in the composition (species, population, community diversity) and the interactions and functional linkages of the different constituents (functional diversity) is taken into account, there is no simple currency for biodiversity offsets.

Principles that regulate common economic markets and those that can be applied to biodiversity trading are fundamentally unequal

Although economic principles will foster few market restrictions and free dynamic markets, some market restrictions will be needed for trading biodiversity to minimise potential risks of further biodiversity loss due to market mechanisms. These restrictions arise from the difficulty to identify ecologically equivalent biodiversity offsets and from the current gaps in the understanding of ecosystem complexity.

PRECAUTIONARY PRINCIPLE FOR BIODIVERSITY MARKETS

Based on the discussion on this topic provided by different authors (Bekessy *et al.*, 2010; Gibbons and Lindenmayer, 2007; McCarthy *et al.*, 2004; Moilanen *et al.*, 2009; Norton, 2009; Walker *et al.*, 2009), several criteria to identify possible 'ecologically equivalent biodiversity offsets' are given below:

a. **Ecologically equivalent biodiversity offsets should be based on type, size, space and time criteria**.
    - **Type.** Ecologically equivalent biodiversity offsets should be carefully identified amongst the same species, community or habitat types. In the case of habitats, native vegetation cannot be traded with non-native vegetation, vegetation types cannot be offset with a different vegetation type, a mature vegetation type cannot be traded with newly-planted vegetation, as considerable uncertainty remains on the long-term development for maturity. In the case of species, ecological equivalence should be based on the functional ecological role they have in the community, as well as the species richness of the community and their dominance/rarity. Biodiversity offsetting that considers out-of-kind ('like for the better' or 'trading up') should be evaluated on a case-by-case basis.
    - **Size.** Ecologically equivalent biodiversity offsets should consider size as a quantitative criterion to identify likely species and habitat offsets. In the case of species, population size should exceed the minimum viable population. In the case of habitats, the overall size of the habitat patch and its shape should be considered as influencing the possible number of habitat-specialist species and habitat-edge species.
    - **Space.** Ecologically equivalent biodiversity offsets should consider that complex spatial networks of interactions existing between populations, communities, habitats and ecosystems. When an element of this network is lost, biodiversity resilience might be affected (Hector and Bagchi, 2007; Loreau *et al.*, 2001). Theoretically, ecologically equivalent offsets must replace the natural capital lost with the establishment of physical infrastructure on the territory. Possible suggestions to decrease the effect of the spatial and functional disruption caused by the loss of a habitat patch within the landscape include: identifying equivalent offsets in the same restricted geographical area; preferring nearby replacement habitat patches over distant ones; concentrating replacement in aggregated sites; and weighting the importance of connectivity to local attributes. The use of these suggestions should be evaluated on a case-by-case basis.

- **Time.** Ecologically equivalent biodiversity offsets should consider permanence in time. Destruction is usually permanent, while protection and restoration of certain habitats can be undertaken only for a definite amount of time, making the long-term conditions of the offset uncertain. Ecologically equivalent biodiversity offsets should be protected/restored/realized before assets are liquidated (see Viewpoint 2 "Growing biodiversity banking").

b. Ecologically equivalent biodiversity offsets **should not be measured with composite, additive indices where one combination of attributes can yield the same score/outcome of another (ecologically different) combination of attributes**.
c. Ecologically equivalent biodiversity offsets **should not be used in case of rare biotopes, habitats of threatened species** or in any other case in which trade adds to an already high risk of extinction or loss. Threatened species and habitats should be considered irreplaceable and not interchangeable aspects of biodiversity.
d. Ecologically equivalent biodiversity offsets **will be directed first according to the conservation of existing habitats, followed by the restoration of damaged, altered habitats** and only with the lowest priority to the creation of new habitats.
e. Ecologically equivalent biodiversity offsets **cannot be approved without a rigorous plan of monitoring and compliance which consider long-term horizons** (more than ten years) overseen by an independent authority working for environmental protection.
f. Ecologically equivalent biodiversity offsets **should take into account the uncertainty of outcomes**. Uncertainty arises when the future value may be less than originally estimated, as a result of which some features of conservation value might completely fail to be established/preserved and/or the success/failure of conservation/restoration might vary amongst several sites.

These criteria, based on the precautionary principle, constitute a general framework for biodiversity markets, but they can also apply to biodiversity PES schemes. As an example, Criterion a highlights the importance of identifying what the component of biodiversity is that PES aims to conserve/restore by qualifying the type, size and proper spatio-temporal scales. Criterion b raises the issue on how to measure biodiversity and the risk of using additive indexes, which do not properly consider the functional role of species in the ecosystem. Criterion c suggests that designing a biodiversity PES programme for the protection of rare habitats and endangered species might be not appropriate because, in this case, the critical situation will require concrete and ad hoc measurements of conservation that might leave little space for negotiation with other needs. Criterion d states that the protection of biodiversity should be considered a priority criterion. For example, PES schemes aimed at the conservation or restoration of natural riparian habitats should be highly preferred over using artificial recreation or monoculture plantations.

Criteria e and f consider the importance of ensuring the long-term compliance of biodiversity protection. Thus, PES programmes should be aimed at conserving existing biodiversity, implementing a plan to monitor the status of biodiversity and environmental compliance of the PES agreement. PES design should also consider precautionary measurements to take into account the uncertainty of outcomes, such as biodiversity levels only partially re-established, longer periods required for a full recovery or unintended leakage.

BUNDLING ECOSYSTEM SERVICES

Most of the current PES schemes are based on the delivery of a single ecosystem service and, thus, they are classified as PES in water, carbon sequestration, biodiversity or landscapes.

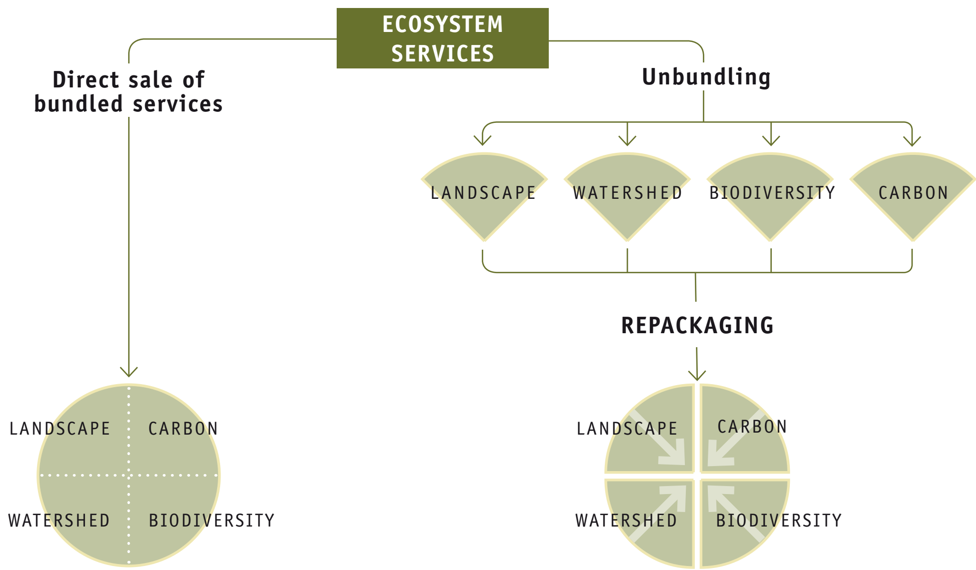
In some instances, instead of considering single ecosystem services, PES projects have considered bundled ecosystem services, for example, in Australia (DSE, 2009), in Costa Rica (Wunscher *et al.*, 2006), in the Danube Delta Region (GEF, 2009), in Colombia, Ecuador and Peru (Goldman *et al.*, 2010), in Kenya (Mwengi, 2008), in Madagascar (Wendland *et al.*, 2010), in Mexico (Muñoz Piña *et al.*, 2008), and the USA (Claassen *et al.*, 2008).

Bundling ecosystem services is commonly understood as a marketing strategy that can be carried out in two different ways: direct sale or the shopping basket (Figure 7). In the direct sale approach, several ecosystem services are sold together as a 'package' and there is no breakdown market analysis of the single ecosystem service components. In the shopping basket approach, however, ecosystem services are initially traded individually and subsequently grouped according to the buyer's needs (Landell-Mills and Porras, 2002). It is clear that the fundamental baseline information needed to sell bundled ecosystem services is the understanding of the relationships occurring amongst ecosystem services in a given location. This implies being able to establish a functional link between agronomic practices and the delivery of different ecosystem services. As an example, in general terms, establishing a riparian buffer can usually enable the delivery of different ecosystem services such as: carbon sequestration, reduction of sedimentation and a decrease of flooding risk. However, this might be a generic and theoretical relationship, while the evaluation of the actual ecosystem in a given location might reveal a more complex network of relationships amongst ecosystem services.

To sell bundled ecosystem services requires an understanding of the relationships amongst ecosystem services in a given location

Ecosystem services interact with each other in a multiple non-linear pathway (Balvanera *et al.*, 2006). There are several typologies of interactions as they can be unidirectional or bidirectional, direct or indirect, with an enhancing or decreasing effect in the provision of the services. To illustrate this conceptual framework, Bennett *et al.* (2009) provide some examples of possible interactions amongst ecosystem services. The level of control of soil erosion can affect water quality (unidirectional-enhancing interaction), while carbon sequestration and tree growth can affect moisture retention (bidirectional-enhancing interaction).

Figure 7
Two approaches to bundling ecosystem services



Adapted from Landell-Mills and Porras, 2002

As ecosystems are complex, ecosystem services will not only interact in a direct way, but their interaction can be mediated by a common driver. A driver is defined as a factor, often directly modified by human management, which affects one or more ecosystem services (Bennett *et al.*, 2009). As an example, wetland restoration (driver) will positively enhance both flood control and water quality (synergy), while fertiliser use (driver) will positively affect crop yield, but negatively affect the provision of water quality (trade-off). Sometimes, a mixed pathway will take place because a driver of change will directly affect one service whose enhanced or decreased provision will, in turn, influence another ecosystem service. For instance, the restoration of riparian wetlands (driver) can enhance flood protection (regulating service), while flood protection can ensure downstream crop production (provisioning service).

Ecosystem services interact with each other in multiple non-linear pathways which affect the provision of those services

In many instances, ecosystem services are affected when ecological principles are not used in ecosystem management. For example, the relationship that exists between afforestation and water supply will vary depending on the tree species used. Usually, the loss of riparian vegetation allows run-off to enter the waterways, carrying with it debris and a variety of other materials, which are likely to decrease water supply and quality (Sweeney *et al.*, 2004).

Interestingly, the same tree species can have opposite effects if planted in areas where it is not native, as opposed to where it is native. When an upper watershed is afforested with native *Eucalyptus* trees, as in the case of New South Wales in Australia, the water supply function can be restored in the ecosystem with the additional advantage of carbon sequestration. In contrast, when *Eucalyptus* trees are introduced to a different ecoregion and are used for the same purpose, as in the case of the Argentinian *pampas*, deep-rooted *Eucalyptus* trees are able to reach groundwater supplies, diminishing the overall water supply (Jackson *et al.*, 2005).

Understanding if ecosystem services interact directly or indirectly through the occurrence of a common driver of change is fundamental for sound management. In human-modified ecosystems, the management of ecosystem services is aimed at increasing synergies and decreasing trade-offs amongst ecosystem services. In situations in which a driver of change strongly affects two different ecosystem services that do not strongly interact with each other, addressing the driver is expected to have an effect on both ecosystem services provision. On the contrary, if the interaction is initiated by a driver, but there is a strong negative and bi-directional interaction between the two ecosystem services (trade-off), managing the driver is unlikely to have any substantial long-term effect (Bennett *et al.*, 2009). As shown in the Millennium Ecosystem Assessment (2005), the benefits of managing ecosystems in a sustainable way exceeded the benefits associated with ecosystem conversion. Thus, in Canada, an intact wetland has a higher economic value than the value obtained if the wetland is converted to intensive farming; in Cameroon, sustainable tropical agroforestry has a higher dollar value per hectare than small-scale farming; similarly, in Cambodia, traditional forest use is more advantageous than unsustainable timber harvest; and, in Thailand, intact mangroves convey ecosystem services for an overall economic value higher than shrimp farming. This is because in the economic evaluation of the total ecosystem value both marketed and non-marketed ecosystem services are considered. Sustainable management of ecosystems should be based on the understanding of possible synergies and trade-offs amongst ecosystem services, which should also be the key information for designing PES schemes.

SPATIAL PATTERNS OF PROVISION OF MULTIPLE ECOSYSTEM SERVICES

Mapping the provision of ecosystem services poses several challenges. The first challenge is linked to the fact that landscapes are heterogeneous with an uneven spatial distribution of goods and services. Within this biophysical variation there is also variation of land use and land management. The second challenge is linked to the fact that different ecosystem services might be characterised by different spatial patterns.

As described by Karoukakis (see Chapter 4 “Cost-effective targeting of PES”), a spatially explicit analysis that compares the occurrence of different ecosystem services can be a useful tool to identify key areas for ecosystem service provision and PES implementation. The simpler way to represent the spatial occurrence of ecosystem services is to associate them to a certain land cover/use. When the study area is spatially delimited and described by a land cover typology, the use of coefficients that express the monetary value of ecosystem services in each cover type might be transferred from other research investigations and used to compute a total ecosystem service value by cover class in the study area (Troy and Wilson, 2006).

Landscapes are heterogeneous and biophysically uneven, as well as having variation in land use and management

This approach, often described as ‘value transfer’ or ‘benefit transfer’, was developed to overcome a lack of data, decrease time and costs for evaluation of ecosystem services provision and to develop global scenarios (Costanza *et al.*, 1997; Troy and Wilson, 2006). However, ‘value transfer’ has been heavily criticised for neglecting potentially important spatial differences that are likely to be found amongst different study areas, different spatial scales and different habitat patches. The assumption that every hectare of a given land cover has a fixed value does not take into account rarity, spatial configuration, size, quality of habitat, type of environmental management, number of resident people, social preferences and motivational attitude towards the preservation of ecosystem services (Tallis and Polasky, 2009). Moreover, the value transfer approach does not consider any change in value of ecosystem services with time (Nelson *et al.*, 2009).

By contrast, the open-source Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) is based on ecological and economic production functions (Nelson and Daily, 2010), rather than benefits transfer. These production functions define how an ecosystem’s structure and function affect the flows and values of ecosystem services. InVEST uses these functions to map the geographic distribution of several ecosystem services, such as water pollution regulation, carbon storage and sequestration, and sediment retention. First, InVEST quantifies biophysical supply (e.g. sediment retention, soil retention capacity), then it maps spatial distribution of ecosystem service (e.g. avoided sedimentation of a reservoir) and, lastly, it can provide economic or social values of the service provided (e.g. avoided cost of sediment removal) (Tallis *et al.*, 2010). To calculate ecosystem service outputs, biophysical outputs are combined with data on demand, such as existing number of beneficiaries and/or the intensity of the demand linked to human activities. The value of the service is estimated through an assessment of cost savings, net present value and other economic methods.

InVEST is scenario driven. In other words, stakeholders can define scenarios for particular land-use/land-cover changes, and trade-offs can be measured through modelling and mapping the provision of multiple ecosystem services under these alternative futures (Nelson *et al.*, 2009). Mapping multiple ecosystem services under present and feasible future conditions makes it possible to assess trade-offs and synergies among ecosystem services and determine how policies and land-use decisions will impact natural capital. However, map comparison has drawbacks. Finding a spatial accordance between two ecosystem services could be difficult to interpret given the approximation introduced by the use of proxy variables to model ecosystem services, uncertainty about accuracy and precision linked to the scales and resolutions of input variables, the occurrence of invisible drivers of changes in the mapping resolution and the different possible measures of spatial congruence, such as overlap, coincidence analysis or correlations (Egoh *et al.*, 2009). In particular, a simple map comparison will not reveal the mechanism or activity through which ecosystem services could be functionally bundled and this lack of information might lead to poor decision-making.

A second drawback is linked to the interdependence between ecological and socio-economic systems. Social factors, such as population density, wealth and increasing economic development, often constitute drivers of change in the ecosystem functioning. Thus, assessing the relationship among multiple ecosystem services with an integrated socio-ecological approach is likely to provide more realistic outputs and the possibility to evaluate existing relationships among ecosystem services against different scenarios of socio-economic changes (Bennett and Balvanera, 2007; Bennett *et al.*, 2009).

Raudsepp-Hearne *et al.* (2010) have suggested a multivariate statistical approach to identify and map bundles of ecosystem services that repeatedly appear together in space amongst municipalities. This heuristic approach, while not investigating the mechanism through which ecosystem services are linked together, can spatially identify situations in which synergies amongst ecosystem services are detected. In this study, only some of the municipalities, characterised by similar levels of crop production, show a severe degradation of other ecosystem services, as measured as soil phosphorous retention, soil organic matter and drinking water quality. This highlights that severe trade-offs between provisioning services (crop production) and regulating services are not always inevitable, but might be driven by policies, environmental awareness and sound management strategies.

In conclusion, spatial explicit modelling tools are often used to generate maps, to create scenarios of change, to provide inputs for discussion amongst stakeholders and to disentangle and understand better bundles of ecosystem services. However, the interpretation of these outputs should always consider the limitations of ecosystem modelling related to the present scarce knowledge of ecosystem functioning (e.g. identification of the threshold at which the functionality of ecosystem services collapse, and understanding of the interactions and feedback loops of ecosystem services amongst multiple spatial and temporal scales).

IMPORTANT ECOLOGICAL CHARACTERISTICS OF PES DESIGN

PES schemes should be designed to reflect important ecological parameters, such as the programme duration, the overall size of area linked to programme, the degree of spatial connectivity and the evaluation of multiple ecosystem services.

- **Duration of the PES programme:** One of the major ecological concerns about PES implementation is the potential disparity between short-term project durations (commonly implemented for about 3-5 years) and the time actually needed to restore and balance the functionality of ecosystems. The time needed to restore ecosystem services will vary according to the biological process involved, such as the vegetation re-growth after reforestation, the time needed for species re-colonisation after local extinction, the time needed to re-adjust population dynamics and community structure after eutrophication and food web modification processes. The conservation and restoration of ecosystem services usually requires a long-term time line. In a review of 89 programmes for the restoration of ecosystem services, the needed time scale ranged from < 5 to 300 years (Rey Benayas *et al.*, 2009). However, the long-term durations of PES programmes are often hampered by the need of a continued flow of financing resources. The long-term duration of PES programmes is also obstructed by the voluntary nature of the agreement in which both the supplier and buyer can withdraw from the programme at any time.
- **Size of the area to be covered by PES:** The overall size of the area that will be linked to the PES programme is clearly a critical ecological parameter; ecological processes are usually affected by biophysical thresholds (Ferraro, 2003). As an example, it is estimated that no substantial increase of water quality could be achieved if agricultural use exceeded 50 percent of the entire watershed (Wang *et al.*, 1997). Similarly, the occurrence of a vegetated buffer is an important factor influencing water purification and the removal of contaminants. The width of the buffer is critical threshold parameters and different widths will be required for the abatement of different contaminants, such as sediments, nitrogen, phosphorus, pathogens and pesticides (Johnson and Buffler, 2008). The size of the area to be covered by PES will influence not only the abiotic properties of ecosystems, but also its biotic components. A quantitative relationship regulates the number of species (species richness) expected in a given habitat patch of a certain size (Stott *et al.*, 1998 versus MacArthur and Wilson, 1967).
- **Spatial connectivity of the area to be covered by PES:** Spatial connectivity is one of the major properties of the landscape that can ensure the long-term survival and persistence of species linked to particular fragmented habitat types. No single population may be able

to guarantee the long-term survival of a given species. Due to demographic stochasticity and the erosion of genetic variability, the smaller the population, the more prone it is to extinction. On the contrary, habitat connectivity will facilitate the establishment of a meta-population structure, constituted by interconnected populations, where emigrants can colonise unoccupied habitat patches or can join a small population and rescue that population from extinction (the ‘rescue effect’).

- **Multiple ecosystem service covered by PES:** Interactions amongst different ecosystems services are what regulate ecosystems. Thus, even if a PES scheme is designed specifically for the delivery of a single ecosystem services a background assessment should evaluate the possible synergies and trade-offs with other ecosystem services.

THE SOCIAL VALUE OF ECOSYSTEM SERVICES

Ecosystem services have a social value because they are natural capital belonging to the whole of society. Having to include many different perspectives and needs, the total value that ecosystem services have for the society is not restricted to direct use, but enlarges to also include indirect use value and non-use value. While the direct value refers to those benefits provided by a direct interaction between people and ecosystems, such as the provision of goods and services and the enjoyment of ecosystem’s beauty through recreational and educational activities, indirect use refers to benefits received indirectly by ecosystem regulating processes. The value of ecosystem services for society can also include non-use value linked to the knowledge that ecosystems continue to exist independently of any possible use (existence value); the awareness that ecosystem services can be enjoyed by other contemporary living individuals (altruistic value); the assurance that ecosystems will be passed on to descendants (bequest value); or the knowledge that ecosystem services will be available for use in the future (option use value) (EFTEC, 2005).

When the total value of ecosystem services is considered it becomes more difficult to assess them in economic terms. Moreover, it can be argued that assigning a monetary value to ecosystem services reduces and distorts their total value. In every society, there are issues that are considered ethically ‘untradeable’, such as human life, friendship, voting or human organs (Vatn, 2000). Ecosystems, as natural resources, are considered as tradable market goods by some people and as having intrinsic, non-quantifiable and non-market value by others. However, even if a direct valuation in economic terms of ecosystem services does not take place, our preferences/choices/actions might reveal that we are indirectly placing a value on them.

ECOLOGICAL SCALES AND INSTITUTIONAL SCALES

Reflecting the true total value of ecosystem services for society is also challenging because the evaluation should include different stakeholders at the local, regional and global scales (see also Chapter 1 “The role of PES in agriculture”). The definition of the scale at which the ecosystem service is supplied implies the specification of the boundaries of the ecosystem that needs to be taken into consideration and this will affect the identification of the institutional scales that need to be involved (Hein *et al.*, 2006).

In general terms, the functioning of a provisioning service will have a direct impact on its direct use by stakeholders at the local and regional scales, the disruption of a regulating service will affect the indirect use by stakeholders also at the regional and global scales, while all stakeholders at all scales will be involved in the alteration of ecosystem services options and non-use values (EFTEC, 2005).

In reality, the situation is more complex. In fact, the decrease of a single service can impact different stakeholders at different scales. As an example, a significant increase in deforestation could determine a long-term reduction in fuelwood provision for local residents, while the increased logging of commercial tree species will affect timber trade and stakeholders at a regional and global scale. The potential of a single ecosystem service to have an impact at local, regional and global scales depends not only on the nature of the service and the occurrence of existing markets for that service or for the goods provided by the service, but also on the cultural backgrounds, societal motivational drivers and personal belief systems. The evaluation of the total value of an ecosystem service is likely to involve different stakeholders at different scales, which can lead to a negotiation process to resolve conflicting views. Hein *et al*. (2006) point out how taking into account different spatial scales can lead to the identifying of varying preferences amongst different stakeholders directly or indirectly involved in the management of the De Wieden wetlands in the Netherlands. The area is one of the most important peatlands in northwestern Europe and is vital for the supply of provisioning services (fish and cut reeds traditionally used for thatched roofs), recreational activities (an estimated 172 456 visitors per year) and the conservation of biodiversity (water birds, butterflies, dragonflies and a population of reintroduced European otter). At the local level, residents are mostly interested in the benefits that they can receive from the use of available resources, such as fishes and reeds, while at national level stakeholders are mainly interested in the potential of this area for biodiversity conservation. This discrepancy also points out the importance of identifying the appropriate institutional level for decision making. A local management plan driven by the preferences of residents will

Valuing an ecosystem service involves different stakeholders at different scales, with negotiations to resolve conflicting views probably not reflect the conservation value of De Wieden at the national and international levels, while a management plan based on national and international regulations could overlook the economic value of provisioning activities for improving local residents' livelihoods. Considering potentially diverse perspectives of stakeholders at different spatial scales will allow the finding of ways to reconcile varied interests and priorities and to make policies and decisions that reflect the total value of ecosystem services for society.

THE POTENTIAL OF PES FOR POVERTY ALLEVIATION

PES was originally conceived as a market tool and not primarily as a tool for poverty reduction. However, the preservation of the ecosystem services has clear connections with the Millennium Developing Goals (MDG), such as eradicating poverty and hunger (MDG 1), improving health and sanitation (MDG 4, 5, 6) and ensuring environmental sustainability (MDG 7). When PES is designed in a way that seeks to express its potential for the achievement of the MDGs and reduce vulnerabilities of the poor, PES becomes equitable and fully expresses its social dimension (Leimona and de Groot, 2010).

Lessons learned from 15 years of PES implementation have point out possible ways to design PES programmes so as to improve their impact on reducing poverty. However, making PES work for the poor requires a shift in perspective and an open attitude to seek ways to reconcile potentially conflicting goals. Adams *et al*. (2004) provide an excellent framework with which to test attitudes between the preservation of ecosystem services and reducing poverty. When considering the conservation of apes in mountain forests (biodiversity ecosystem service) and poverty, assuming that poverty does not play a role in the dramatic reduction of ape populations in the Congo basin, one would probably simply advocate for strictly-enforced protected areas. On the other hand, considering the poverty conditions in the area as a critical constraint on the success of ape conservation, the implementation of programmes that seek cooperation and discourage people living around such parks from trespassing or hunting in the protected area would be promoted. However, if not only poverty is considered having an effect on biodiversity conservation, but also that ape conservation programmes have a potentially negative impact on poverty, one would try to fully compensate resident people for the associated opportunity costs of the park and turn the interests of local communities to preserve rather than exploit vulnerable ape populations.

PES was originally conceived as a market tool and not primarily as a tool for poverty reduction

It is clear that the last attitude fosters the design of PES programmes, particularly in terms of increasing their potential for poverty alleviation. In this respect, particular attention should be focused on: property rights allocation, abetment of transaction costs, occurrence of a trustworthy intermediate agent, and fair and participatory establishment of the compensation of forgone alternative land uses. These elements in the PES design will enhance the eligibility, interest and ability of poor households to participate in PES programmes (Pagiola *et al.*, 2005).

The most important factor that can prevent the participation of poor people is a lack of land property rights. Thus, when PES programmes promote a clear legal definition of land tenure, this is already an important step in the direction of poverty alleviation as resources and property rights become defined for present and future generations. On the other hand, the poor often own very small parcels of land which will have a limited impact on ecosystem services. Thus, if a simple criterion of additionality is used, the inclusion of poor farmers will undermine a credible demonstration of additionality. In this case, implementing a PES programme at the community level can overcome such constraints and reduce the transaction costs of contracting single individuals. Another advantage of implementing PES programmes at the community level is the possibility of paying rewards to the community in terms of improvements of education or sanitation (construction of school, hospitals, etc.). These non-financial incentives can significantly contribute to improve local livelihoods, especially of landless people who will indirectly benefit from PES initiatives (see also Chapter 6 "Landscape labelling approaches to PES: Bundling services, products and stewards").

Proper PES design can enhance the eligibility, interest and ability of poor households to participate in PES schemes

Another barrier is often represented by the initial cost that poor farmers face when adopting land-use or agronomic practices fostered by the PES programme. Most PES projects consider an initial disbursement to cover these establishment costs (Pagiola *et al.*, 2007) and partly overcome a financial constraint of poor landholders to participate.

Often poor people are also constrained in their ability to participate in PES due to a lack of supportive regulations and/or a lack of skills, knowledge and adequate social network. In this case, the role of an intermediate agent that is trusted and considered reliable by local people is fundamental to representing the interests of poor communities and mediating their perspectives with that of different stakeholders.

Last but not least, poor landholders might not be interested in joining PES programmes because the restriction of future land-use options can be perceived as a too high opportunity cost. Thus, it is important that PES programmes enhance social dialogue and participatory approaches amongst stakeholders to reflect the true opportunity costs perceived by local people. Moreover, PES design should be built with a certain amount of flexibility to be able to adjust to the potential change of opportunity costs over the years.

PROMOTING COMMUNITY PARTICIPATION IN PES PROGRAMMES

PES programmes that aim to promote community participation and enforcement should enhance social dialogue to allow the formation of societal and community preferences, avoid and monitor the surge of conflict or strategic behaviour, be implemented according criteria of equity and social justice and foster collective action.

Societal preferences

The neoclassical economic framework is based on two main unrealistic assumptions: (a) that individual preferences generally remain fixed under all circumstances, and (b) that societal preferences can be expressed as the sum of individual preferences. In reality though, individual preferences change with time and under the influence of education, advertising, variations in abundance and scarcity of goods and services, changing cultural assumptions and specific social and environmental contexts. Moreover, single individuals can have plural identities, showing diverse behaviours in different social contexts, which do not necessarily reflect rational consumer choices (Chee, 2004). The preferences and attitudes of individuals towards public goods and ecosystem services are highly influenced by socio-cultural contexts, learning, knowledge-sharing and social discourse. Thus, participatory processes are essential incubators that allow the formation of social preferences, seed motivational drivers at individual and community level and set the basis for a consensus and collective action (see also Chapter 5 “Social and cultural drivers behind the success of PES”).

Participatory processes allow the formation of social preferences, seed motivational drivers and foster collective action

Conflicts and strategic behaviour

Conflicts often arise from a sense of social injustice. Clearly, the establishment of a PES scheme can increase the potential for social conflicts. Conflicts can arise amongst participants in PES programmes and/or between participants and outsiders. The two primary controversial issues are the criteria for property rights allocation and the criteria for defining the opportunity costs and compensation.

Often an indication of a certain level of social conflict is given by the appearance of strategic behaviour. Strategic behaviour is intended to influence the market environment in which it operates to turn the markets to the advantage of the individuals adopting them (see also Chapter 4 “Cost-effective targeting of PES”). In PES schemes, strategic behaviour mainly refers to market operation and speculation to increase the value of the land, ad hoc changes in land use to be eligible for present or future PES schemes, strategic immigration to the area where PES programmes are forthcoming and strategic behaviour in contingent evaluation and bidding rounds for determinations of opportunity costs of their lands (Ferraro, 2001; Ferraro and Pattanayak, 2006).

Criteria of equity and social justice

Equity and social justice are the basis to promoting a sense of community and collective action. PES should be carefully designed if it aims to reflect equity and social justice. In fact, criteria of additionality and economic efficiency may not reflect criteria of fairness and justice.

As an example, Salzman (2005) discusses a virtual scenario in which two farmers own adjacent properties on a slope next to a small river flowing into a reservoir. While five years ago the first farmer, having some environmental concerns, fenced his property to avoid soil erosion and the run-off of nutrients into the stream, the second farmer continued business as usual. If a PES project is set in the area to improve the water quality of the reservoir, an incentive to improve agro-ecological practices is likely to be offered to second and not to the first farmer.

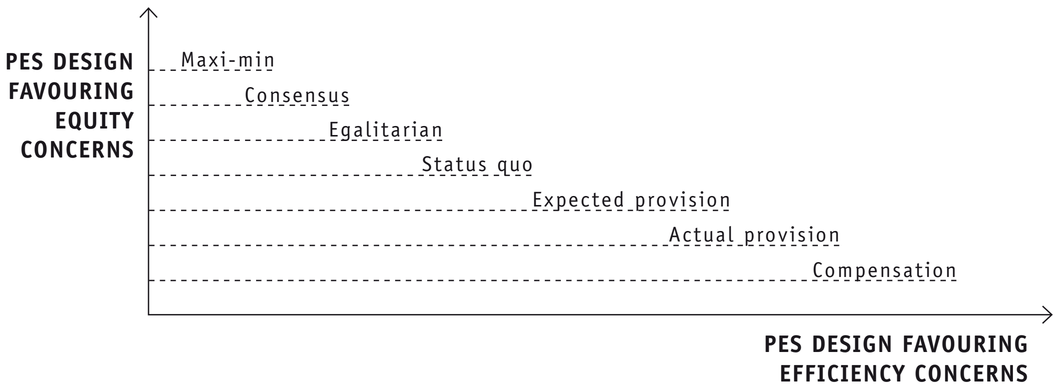
PES schemes reward improvements, not previous efforts made; this rationale often does not promote equity and social justice

In fact, PES schemes are commonly designed to reward an improvement in ecosystem service provision (Salzman, 2005). Under the additionality criterion, PES should reward only additional improvements and not those that would have been adopted anyway. Additionality is considered a pre-requisite to achieve economic efficiency, but this often does not consider consequences on equity and social justice. To overcome this gap and credit the landowners for the ecosystem service provision they have done prior to participation in the programme, some projects have made an initial disbursement, which was not linked to subsequent farmers enrolling into the PES programme (Rios and Pagiola, 2009).

Economic efficiency and equity and social justice can be considered two independent principles that stand on two orthogonal axes and payments can be made according to different criterion that reflect the various degrees and mixtures of economic efficiency and equity (see Figure 8).

At one extreme, payments can be set to optimise economic efficiency and be strictly tailored to the opportunity costs of the different landowners (compensation criterion); at the other extreme, payments can be set to maximise the net benefit to the poorest landowner (maxi-min criterion). Between these two extremes, there are also intermediate solutions (see Table 4). The choice of a given criterion will highly affect the overall performance of a PES scheme. As an example, criteria for calibrated payments that are decided by community agreement (consensus criterion) are likely to promote cooperation, social stability and collective action.

Figure 8
PES design and different emphasis on equity and efficiency criteria



Adapted from Pascual *et al.*, 2009

Table 4
PES design and different fairness criteria

Fairness criterion	Design implications
Maxi-min	Payments aim to maximise the net benefit to the poorest landholders, even at a cost efficiency loss. Payments are differentiated according to the income of providers.
Consensus	Design should promote group decision-making processes to distribute the available funds in a consensus basis. The criteria for payment differentiation are decided by consensus.
Egalitarian	Design should distribute the fund equally among all the providers (per unit of land area, for example), independently of the level and cost of ES provision. Payments are not differentiated.
Status quo	Payments should maintain the previous level of relative distribution of income among providers. Payments are differentiated according to the impact on income equality.
Expected provision	Payments to landholders depend on the expected level of provision of services for a given land use. Payments are differentiated according to the expected provision of ES. These payments compensate landholders to particular land-use changes or practices expected to enhance the provision of ES.
Actual position	The allocation of funds among landowners corresponds to the actual outcome level of provision of ES. Payments are differentiated according to the actual provision of ES.
Compensation	Payments should compensate landholders for the forgone benefits related to the provision of ES. Payments are differentiated according to the cost of provision.

Adapted from Pascual *et al.*, 2009

Collective action

While PES originates as agreements contracted between several single landholders, many lessons learned suggest a potential to shift the contractual agreement of PES from the individual to the community.

Engaging in PES schemes with single private landowners has several disadvantages, including high transaction costs, the reinforcement of competition amongst potentially interested participants in the PES programme, the difficulty of revealing the true opportunity costs in such competitive social contexts and the likelihood of some landholders being against the programme and, thus, acting as 'free riders' or opponents to the PES programme.

On the other hand, collective action at the community level will benefit the provision of several ecosystem services. In some instances, ecosystem services have important threshold effects, meaning that if not adopted on a large enough area, the benefits are not realised at all (e.g. the protection of the habitat for some endangered species will be effective only if the area is large enough for a viable resident population). In other instances, ecosystem services can be disrupted if proper management is not adopted by all community members (e.g. a single source of pollution can make the efforts of a large number of actors meaningless).

Lessons learned suggest shifting the contractual agreements of PES from the individual to the community

Collective action can provide several advantages. It might be important in creating collective opposition against unwanted institutional change. In particular, a cohesive community can influence land property allocation or a community residing on public land can foster community user rights (Wunder *et al.*, 2008). Collective action can also strengthen the bargaining power of smallholders, reduce transaction costs, increase cooperation and have greater potential to set up PES schemes that require coordination among neighbouring landowners (Goldman *et al.*, 2007; Parkhurst *et al.*, 2002). In particular, Goldman (2010) describes how the spatial configuration (placement) and composition (type) of native vegetation on agricultural landscapes can be critical to enhancing the provision of different ecosystem services (Viewpoint 3 "PES design: Inducing cooperation for landscape-scale ecosystem services management").

The main difficulty in generating collective action is that landscapes, by their very nature, are heterogeneous and, thus, not all land or landholders are equally important in the delivery of ecosystem services. As an example, certain areas which include stream banks, steep hillsides and wetlands may need to be managed more carefully than other areas. Furthermore, not all watersheds have the same importance; those upstream of major cities, industries, hydroelectric facilities or other critical water users are likely to receive greater attention.

This implies that even in community-based PES schemes, a calibrated differentiation amongst community members is most likely to be necessary to reflect the true opportunity costs. However, if this evaluation is assessed through the consensus of the community, the contractual agreement of PES could be still made with the whole community and part of the reward could be paid as infrastructure (i.e. non-financial remuneration) for the improvement of living conditions of all the community members.

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GROWING BIODIVERSITY BANKING

Sarah A. Bekessy[1], *Brendan A. Wintle*[2], *David B. Lindenmayer*[3], *Michael A. Mccarthy*[2], *Mark Colyvan*[4], *Mark A. Burgman*[2] *and Hugh P. Possingham*[5]

[1]School of Global Studies, Social Science and Planning, RMIT University, Melbourne, Australia
[2]School of Botany, The University of Melbourne, Melbourne, Australia
[3]Fenner School of Environment and Society, Australian National University, Canberra, Australia
[4]School of Philosophical and Historical Inquiry, University of Sydney, Sydney, Australia
[5]The Ecology Centre, The University of Queensland, St. Lucia, Australia

Biodiversity banking and vegetation offset schemes are now applied in countries around the world in an attempt to halt ongoing vegetation loss in already heavily altered landscapes (Fox and Nino-Murcia, 2005). Under these schemes, proponents of a development involving clearance or alteration to vegetation are required to provide an offset of an equivalent or better biodiversity value, evaluated using a biodiversity value metric. However, offsetting vegetation destruction to mitigate environmental damage will unquestionably result in further loss of biodiversity unless a more rigorous scientific approach is adopted (Bekessy *et al.*, 2010).

ALLOWING THE PROTECTION OF EXISTING ASSETS AS AN OFFSET WILL DEPLETE BIODIVERSITY

Many biodiversity banking schemes allow vegetation clearance to be offset by the protection of existing vegetation through changes in tenure or security arrangements, rather than requiring revegetation of cleared areas. This will result in a net loss of habitat. In the best-case scenario, when the offset site is protected in perpetuity and managed so that its condition improves over time, there is still a net loss of habitat. However, many biodiversity banking schemes include ambiguous responsibilities for ongoing protection and management of offsets, which many lead to even greater losses of habitat in the landscape.

UNCERTAINTY PRECLUDES THE PROMISE OF FUTURE REVEGETATED HABITAT AS A NET-GAIN OPTION

The uncertainties surrounding revegetation success are very high (Hynes *et al.*, 2004) and multipliers to account for uncertainties are likely to be unworkably large (Moilanen *et al.*, 2008). Furthermore, time lags in the availability of habitat may result in populations dropping below a minimum viable population size (Shaffer, 1981).

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The unacceptably high level of risk to the environment of trading immediate loss of existing habitat against uncertain future gains through revegetation means that the value of offsets should be realised before assets are liquidated.

THE BIODIVERSITY BANK AS A SAVINGS BANK

It is proposed that for biodiversity banking to provide genuine net-gain outcomes, biodiversity assets must be banked for the future and trading allowed only once it can be demonstrated that assets have matured (Bekessy *et al.*, 2010). The value of 'saved' biodiversity assets should be demonstrated before they can be made available to offset loss of vegetation elsewhere. Mature vegetation could be sold to a party interested in clearing an equivalent amount and quality of vegetation. Alternatively, a market could be established for buying and selling banked biodiversity (i.e. habitat created above and beyond 'duty of care'). A few other considerations include:

- The currency of trade must reflect ecological realities, including irreplaceability (Pressey *et al.*, 1994) and the dynamic nature of landscapes;
- Responsibility for maintaining and protecting offsets must be identified;
- Implementation must be closely regulated and legally enforceable (Bekessy *et al.*, 2010).

USING CARBON INVESTMENT TO GROW THE BIODIVERSITY BANK

If correctly harnessed, the power of carbon initiatives could fuel the biodiversity savings bank (Bekessy and Wintle, 2008). An important step will be to allow investors to simultaneously accrue carbon and biodiversity credits from the one parcel of land.

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Previous page:

↖ Replacing old-growth forests with plantations negatively affects ecosystem services, especially carbon sequestration and biodiversity.

Current pages (from left to right):

→ Deforested slopes can create a disruption in water and soil ecosystem service delivery.

→ While offsets can include the rehabilitation of logged forests, ecological restoration is often very long and difficult, so conservation should be the priority.

→ Land management practices can impact carbon emissions, so changes in emission regimes can be also sold as an offset.

CONCLUSION

Biobanking may have appeal as an elegant economic instrument for balancing economic growth with biodiversity conservation. However, the purpose is dubious if it fails to deliver real benefits for biodiversity and may, in effect, reduce pressure on developers to avoid harm. The extinction debt in many parts of the world from past clearance means that we need vegetation policies that aim to achieve net gain in the landscape. The only way to achieve this through offsetting schemes is if the biodiversity bank is established as a genuine savings bank.

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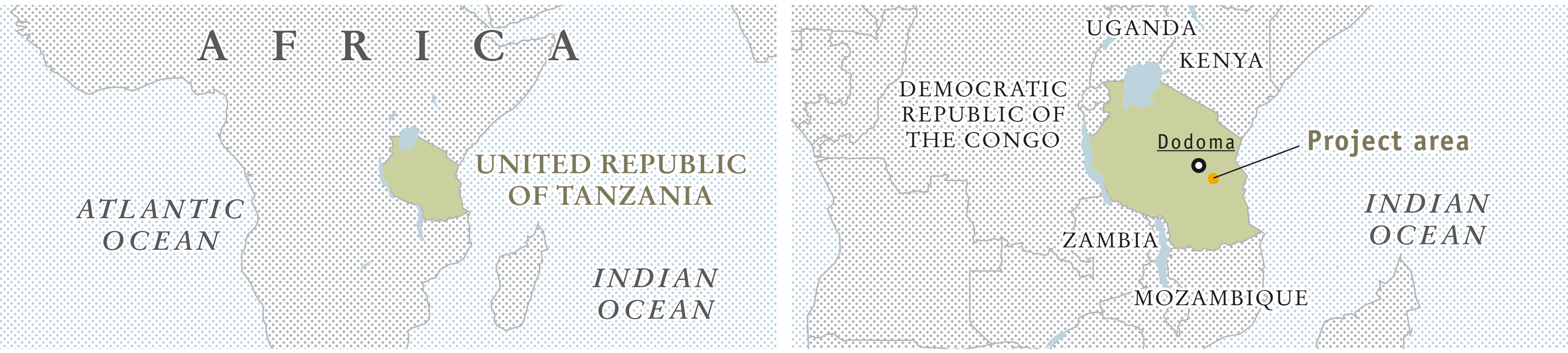
PES IN THE RUVU WATERSHED OF THE ULUGURU MOUNTAINS, TANZANIA

Dosteus Lopa[1] *and Rohit Jindal*[2]
[1]CARE International, Dar-es-Salaam, Tanzania
[2]Michigan State University, East Lansing, Minnesota, USA



The Uluguru Mountains are a range in eastern Tanzania that blocks the moisture coming from the Indian Ocean. Consequently, they are characterised by wet slopes, where the overall annual precipitation on the east-facing slopes exceeds 2 000 mm. Rainfall is captured in a complex network of streams that join to form the Ruvu River, which supplies water to over four million people in Dar-es-Salaam and to the major industries of Tanzania. About 150 000 people live in the Uluguru Mountains in about 50 villages situated on the edge of the forested areas.

In 2007, a hydrological assessment by CARE-WWF revealed an overall decrease of water quality with an average increase of five NTUs (Nephelometric Turbidity Units) per year, indicating a dramatic increase in sediment loading into the river. At the same time, significant fluctuations have been recorded in the annual volume flow of the Ruvu River due to variations in the precipitation regime, as well as to the runoff and overall decrease of the storage capacity of the river's tributaries. As a consequence, downstream water treatments are needed due to high level of siltation of the Ruvu River and often downstream water supply needs to be rationed. The restoration of the Ruvu's hydrologic services is mainly linked to improved upstream land-use management, which is strictly linked to poverty alleviation and livelihood improvements of the people inhabiting this region with a very high population density.

Thirty-one percent of the population of the Uluguru live on less than one dollar (USD) per day, with subsistence farming of very small agricultural plots that are managed with slash-and-burn practices. Land fragmentation is extremely high and aggravates food security. According to the CARE-WWF investigation (2007), 86 percent of the farmers in Kibungo-Juu own no more than two hectares of land. Productivity of such small agricultural plots is very low due to low soil fertility (e.g. on average, about 200 kg of maize per acre) and financial constraints in implementing practices to counteract the continuous loss of soil and nutrients by erosion and runoff.

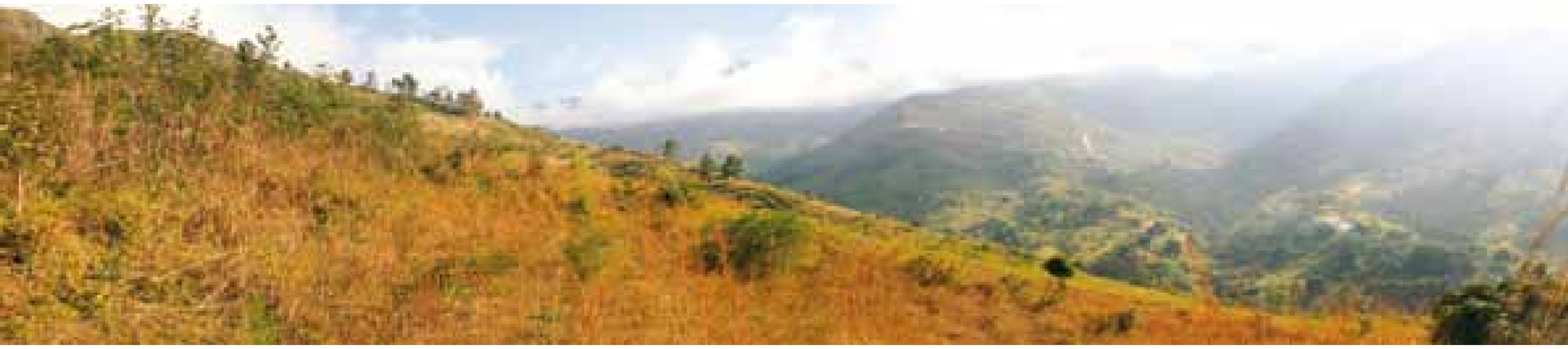
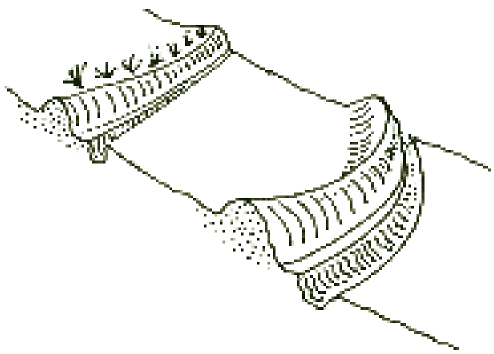
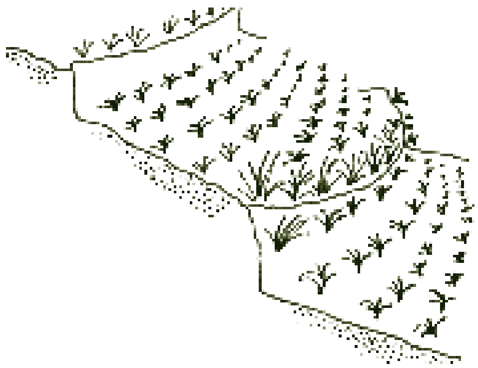


Figure 9
Newly established traditional terrace (*fanya juu*)



Source: IIRR, 2008

Figure 10
Traditional terrace (*fanya juu*) after 5 years



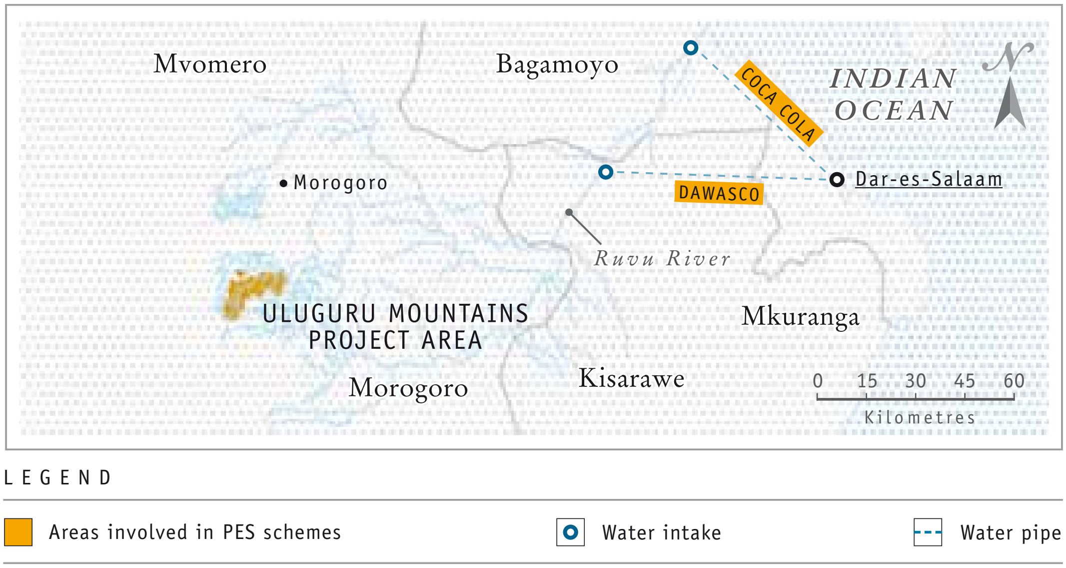
Source: IIRR, 2008

In the subcatchment of the Mfizigo River, a joint CARE-WWF Programme (2006-2011) promoted a PES scheme between the downstream buyers (the industrial Water Supply and Sewerage Corporation [DAWASCO] and Coca Cola Kwanza Ltd.) and the upstream sellers (currently about 265 farmers are engaged) from the Lukenge, Kibungo, Lanzi, Dimilo and Nyingwa villages (Figure 11).

Farmers received payment for the adoption of agriculture practices aimed at controlling runoff and soil erosion, while improving their crop production. A combined approach is being implemented that includes structural (bench terraces and *fanya* terraces) (Figure 9 and 10), vegetative (reforestation, agroforestry and grass strips) and agronomic measures (intercropping crops with fruit trees, mulching and fertilising with animal manure) to limit runoff, combat soil erosion, and increase soil moisture and productivity.

Figure 11

Location of the area of PES scheme implementation and locations of the two main companies paying for increased water quality and quantity of the Ruvu River



Adapted from original map by Heri Kayeyey Masudi (Sokoine University of Agriculture)

Payments are allocated according to how many hectares of land are converted and the type of agricultural and/or land-use practice adopted. The estimated costs of the adoption of these practices (Table 5) were evaluated by CARE-WWF upon consultation with discussion groups and village assemblies and an evaluation of economic returns provided by maize, beans, cassava, rice and bananas, the most common crops in the Uluguru area (Lopa, 2010).

An auction carried out by PRESA in the Kinole area and sub-catchment of the Mbezi River (March 2009) also provided additional information on the estimated opportunity costs related to reforestation activities. The auction involved over 300 participants belonging to ten different

Table 5

Opportunity costs and payments received by farmers for soil erosion control practices

Structural and vegetative agronomic practice to control runoff and soil erosion	% of land that will not be cultivated due to the adoption of a particular agronomic practice	First year opportunity cost (Tsh./ha)	First year labour cost (Tsh./ha)	First year total cost (Tsh./ha)
Bench terraces	100%	160 000	210 000	370 000
Reforestation	100%	160 000	75 000	235 000
Riparian restoration	100%	160 000	12 000	172 000
Fanya juu	20%	32 000	155 610	187 610
Agroforestry	17%	27 200	13 500	40 700
Grass stripping	17%	27 200	13 500	40 700
Pineapple contour farming	14%	22 400	18 000	40 400

Tsh. = Tanzanian shillings
Source: CARE-WWF, 2008

settlements and revealed the costs perceived by the farmers for changing their land use from seasonal cropping to woodlots using different types of autochthonous trees. The mean estimated cost of planting 400 trees over one hectare (at a spacing of 5x5 m) and for protecting trees for at least three years was of about Tsh. 240 000. During these three years, farmers were responsible for looking after their trees, although they were free to grow crops between the trees. In a first bidding round, the cost of planting 40 *Khaya anthoteca* trees (an indigenous timber species) and 40 *Tectona grandis* trees (teak, a slow growing tree that is popular among local farmers for its valuable timber) was estimated, while in a second bidding round, a mix of species of 40 *Khaya anthoteca* trees and 40 *Faidherbia albida* trees (an indigenous tree that can grow among field crops as it sheds its leaves during the rainy season and provides firewood

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Previous pages:

↖ The Uluguru Mountains in eastern Tanzania are characterised by an extremely variable vegetation ranging from coastal to montane and upper montane forest types.

Current pages (from left to right):

→ Stakeholder consultation with a local community in the Uluguru Mountains led by representatives of CARE and the World Agroforestry Centre.

→ Traditional agricultural landscape with the fanya juu terracing system supports soil conservation.

→ The devastating effects of soil loss from runoff on degraded land.

and traditional medicine). Despite the species mix used, the opportunity costs of these two bidding rounds were very similar (Jindal, 2010).

The case study of PES in the Uluguru Mountains shows how estimating the opportunity costs is a key factor in the design of PES schemes to ensure farmers participation. Long-term involvement of farmers is also necessary to meet the time scale requirements to restore the functionality of ecosystem processes.

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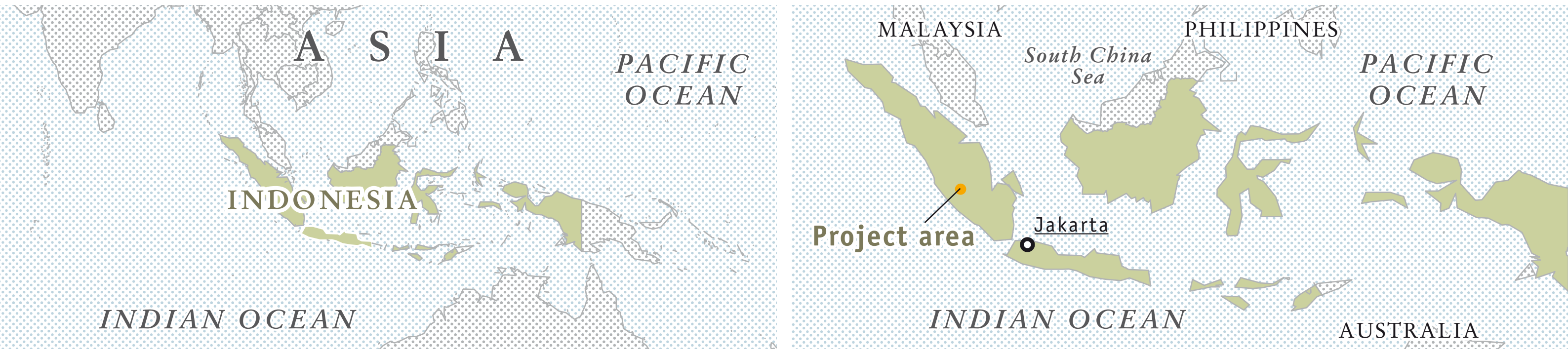
RUBBER AGROFORESTRY AND PES FOR PRESERVATION OF BIODIVERSITY IN BUNGO DISTRICT, SUMATRA

Laxman Joshi[1], *Rachman Pasha*[2], *Elok Mulyoutami*[2] *and Hendrien Beukema*[3]
[1]International Centre for Integrated Mountain Development (ICIMOD), Kathmandu, Nepal
[2]World Agroforestry Centre (ICRAF), Nairobi, Kenya
[3]University of Groningen, Groningen, The Netherlands

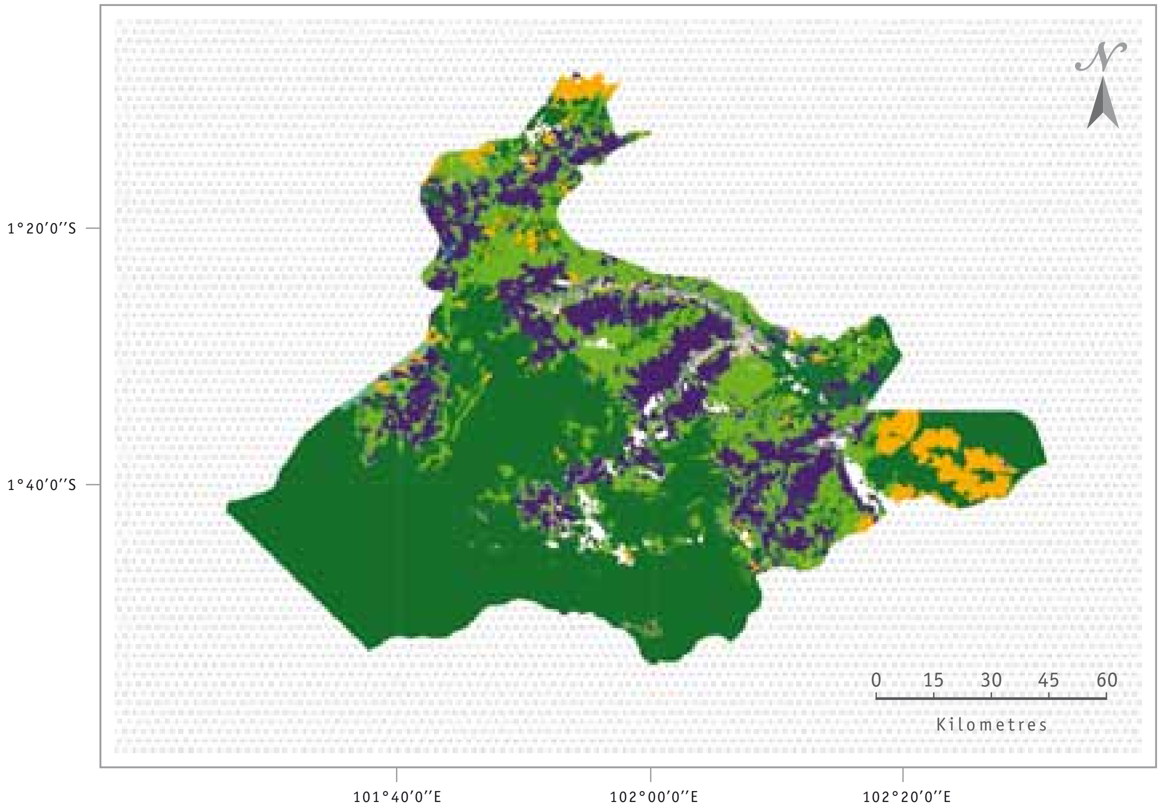


The introduction of the rubber tree (*Hevea brasiliensis*), naturally found in the floodplains forests along the Amazon River, began in Indonesia in the second half of the 19th century. In Sumatra and Borneo, rubber cultivation, initially restricted along rivers with good accessibility, rapidly spread to even relatively remote areas in the country. Currently, Indonesia is the world's second largest gum exporter with an overall rubber area of 3.5 million hectares. More than one million households depend on rubber-generating income in Indonesia, as 83 percent of the rubber cultivation area is constituted by smallholder rubber agroforestry systems (Wibawa *et al*., 2005).

Bungo district, located in the western area of the Jambi Province, the third most important Indonesian province for rubber production, is surrounded by three national parks: Kerinci Seblat, Bukit Dua Belas and Bukit Tiga Puluh. The district has been severely deforested (60 percent forest loss) and forests have been replaced by rubber and oil palm plantations, as well as other agricultural land uses. In particular, from the late 1980s, an increased spread in oil plantation cultivation has led to the additional loss of native trees and simplification of the agro-ecological landscape (Fentreine *et al*., 2010). A remote sensing study showed that in 1998 the remaining forests, mostly located on the Barisan range, covered only 28 percent of Bungo district, while in the area occupied by jungle rubber has decreased from 17 percent (1988) to 11 percent (2008) due to a parallel increase in monoculture covering from 23 percent (1988) to 49 percent (2008) of the district area (Ekadinata *et al*., 2010) (Figure 12 and 13).

In Bungo district, rubber is cultivated in monoculture systems, as well as in more complex rubber agroforestry systems. A rubber agroforest usually starts from slashing a forest plot (either primary or secondary forest) or an old rubber garden, followed by burning the felled trees during the dry season. For the first one to two years, rubber seedlings are grown with rice and other annual crops. When the rubber trees begin to shade annual crops, the plots are left 'fallow' and the native vegetation regenerates. Non-rubber trees are regularly removed or kept below the level of rubber trees and periodic weeding is done around the rubber saplings. The rubber trees reach maturity in seven to ten years, at which time the farmers begin tapping (Joshi *et al*.,

Figure 12
Land cover of Bungo district in 1988



LEGEND

- Forest
- Rubber forest
- Rubber plantation
- Oilpalm plantation
- Rice paddy
- Shrub
- Settlement
- Water body

Adapted from original map by Andree Ekadinata (ICRAF)

Current pages (from left to right):

→ Surroundings of Lubuk Beringin, the first village granted with the legal right (*hutan desa*) by the Indonesian Government to manage state forests for their own prosperity.

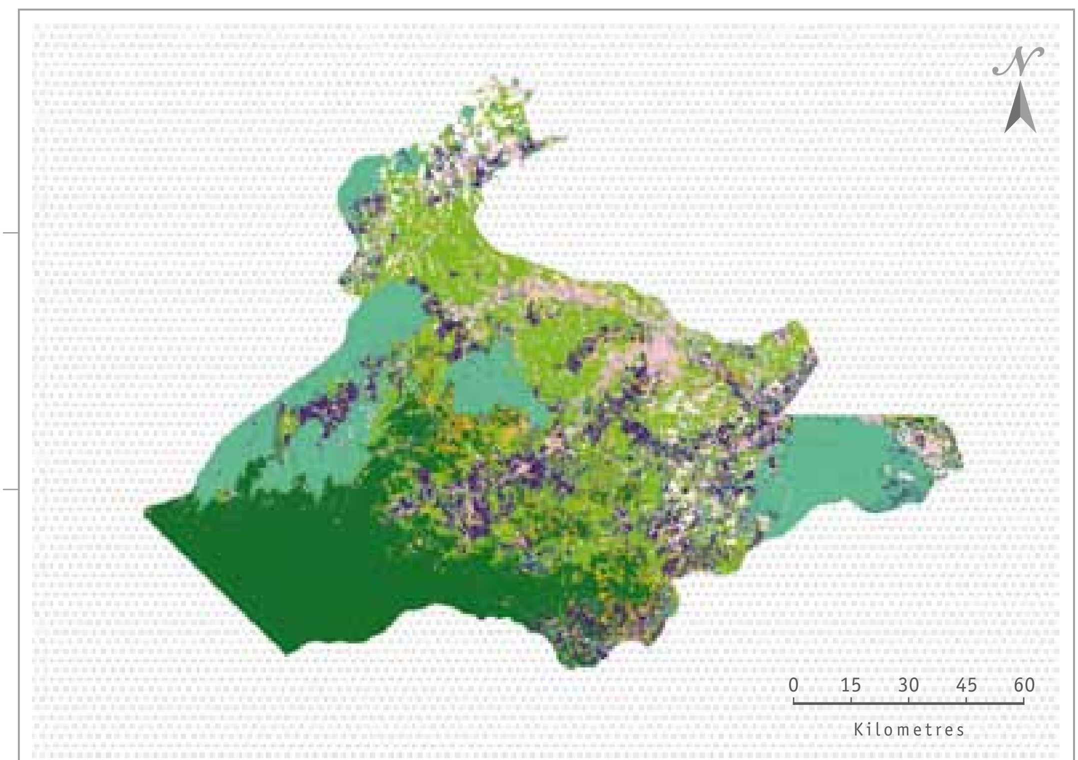
→ View of the forested area designated for community forestry permits, which could help meet forest management targets and livelihood interests of local villages.

→ Rubber jungle, a traditional agroforestry practice that mixes jungle plants among rubber trees.

→ Example of jungle rubber bordering a rice paddy.

Figure 13

Land cover of Bungo district in 2008



LEGEND

Forest	Oilpalm plantation	Settlement
Rubber forest	Rice paddy	Water body
Rubber plantation	Shrub	

Adapted from original map by Andree Ekadinata (ICRAF)

2003; Wibawa *et al.*, 2005). These traditional rubber gardens are complex in structure. Gradually over time rubber trees die due to natural causes and other native species begin to become more dominant. The latex productivity in these gardens, thus, gradually declines. About 25-40 years after planting, when tapping is no longer economical, all the trees are felled and the plot is cleared for replanting. However, some farmers plant rubber seedlings in the gaps caused by the death of rubber and non-rubber trees; this gap-planting, locally known as *sisipan*, leads to unevenly aged rubber trees when carried out over multiple years. The rubber productivity period can be prolonged using the *sisipan* technique, but the *sisipan* plots are never as productive as normal rubber gardens. Compared to slash-and-burn, however, the *sisipan* practice is less labour intensive and does not require much capital investment. It also allows a reduced but continuous income from the plot (Joshi *et al.*, 2002; Wibawa *et al.*, 2005); hence, it is practised mostly by poor farmers in less accessible areas. The biodiversity inside such *sisipan* plots is normally very high, comparable to surrounding forests both in structure and function as large trees and naturally regenerating vegetation is retained in the plots. These plots become 'very complex rubber agroforests' that are often referred to as 'jungle rubber'.

In 2004, ICRAF initiated a PES pilot project in Bungo district (Jambi province) to develop a reward mechanism in order to conserve the rich biodiversity inside the complex rubber agroforests.

In general terms, quantifying biodiversity in jungle rubber is methodologically quite challenging as the potential occurrence of many confounding variables and the high variability found amongst jungle rubber gardens would require a large number of sampling units. In fact, in the Jambi region, rubber cultivation is composed of a mosaic of small jungle rubber gardens at different development stages, rubber densities and management practices. Potential factors that influence the species number (α diversity) and the rate of change in species composition (β diversity) are the plot size, the history and management of the plot and the surrounding landscape, the geographic location of the jungle rubber garden, the elevation, and the adjacency to forest remnants, to other rubber jungles or the influence of an agricultural matrix (Beukema *et al.*,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Current pages (from left to right):

→The economic boom in palm oil since the 1980s has seen millions of hectares of community forests in Sumatra converted into oil palm plantations.

→Oil palm is much more profitable for smallholders than rice production and is highly competitive with rubber.

→In Bungo, rubber cultivation is done in a mosaic of small rubber jungle plots interspersed with other crop fields, such as rice paddies.

→Rice paddies near Lubuk Beringin village are an important livelihood source for villagers in Bungo.

2007; Wibawa *et al.*, 2005). In addition, extensive biodiversity surveys in tropical ecosystems are very challenging due to the high density of species (e.g. 100 vascular plant species in 0.02 ha of jungle rubber) and the difficult and time-consuming task of species identification (Gillison *et al.*, 2000*b*).

A study of the available published and unpublished investigations conducted in the 1990s on €and β diversity recorded in primary forest, jungle rubber and rubber monoculture plantations revealed that jungle rubber had a much lower number of epiphytic pteridophyte and tree species, a similar number of bird species, and a higher number of terrestrial pteridophyte species than primary forest (Beukema *et al.*, 2007). The lower number of epiphytic pteridophyte species may be due to the fact that many epiphytes depend on later successional stages of forest and may not have had enough time to establish and reproduce. Thus, for some species, even a 40-year-old jungle rubber garden might be too young to serve as a suitable habitat.

The lower richness of tree species recorded in jungle rubber (Figure 14) may also be explained by the fact that jungle rubber is a type of secondary forest, where late-successional tree species may not have established yet. Selective species removal by the farmer is another important factor.

Although the total number of bird species in jungle rubber and primary forest (Figure 15) was similar, the number of forest-specialist birds was much lower in jungle rubber.

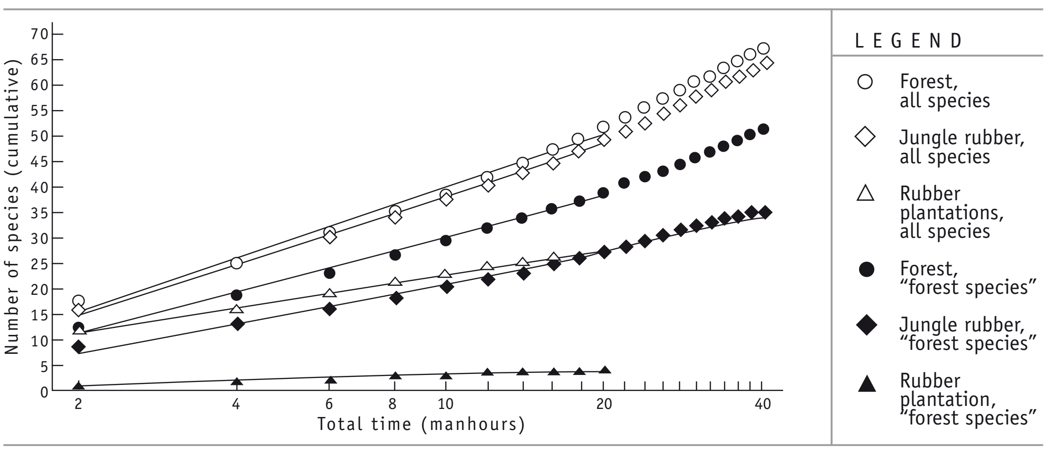
The same was true for terrestrial pteridophytes (Figure 16): for a subset of forest species, the number of species found was much lower in jungle rubber than in primary forest (Beukema *et al*. 2007).

RUPES also carried out rapid biodiversity assessments in Bungo district and found that of a total of 971 tree species recorded inside jungle rubber gardens (77 analysed plots), 376 tree species were found both in jungle rubber gardens and natural forest patches (31 analysed plots). Complex rubber agroforests also harbour a fair number of mammals species (n=37) compared to the number found in the surrounding national parks (n=85). Of these 37 mammals species, nine are endangered species under CITES criteria (ICRAF, n.d.).

Figure 14

Species-accumulation curves for individual trees of DBH over 10 cm, for 3.2 ha of primary forest (Laumonier, 1997, dots) and 3.2 ha of jungle rubber (Hardiwinoto *et al.*, 1999; diamonds).

Open diamonds: all trees including rubber trees. Filled diamonds: rubber trees excluded from the jungle rubber data.



Adapted from Figure 6 in Beukema *et al.*, 2007: 227

The biodiversity assessments indicated that complex rubber agroforests in Bungo not only represents secondary habitats/refuges for forest species, but they are also important connectors amongst remaining fragmented forest patches. According to the landscape configuration, complex rubber agroforests can constitute a series of stepping stones or more continuous corridors (van Noordwijk, 2005).

At the community level, the RUPES project initiated a number of activities aimed to assess the strengths, weaknesses, threats and opportunities of traditional rubber cultivation that can maintain rich biodiversity. Local perception and needs were assessed through consultations and research. Activities to enhance the awareness of the local communities about the value of their traditional system for biodiversity conservation were implemented. Communities of Letung, Sangi, Mengkuang Besar, Mengkuang Kecil and Lubuk Beringin villages agreed to retain their complex

Current pages (from left to right):

→Natural rubber comes from the milky latex found in the bark of rubber trees.

→Tapping involves extracting latex from a rubber tree by shearing off a thin layer of bark in downward half spiral on the tree trunk.

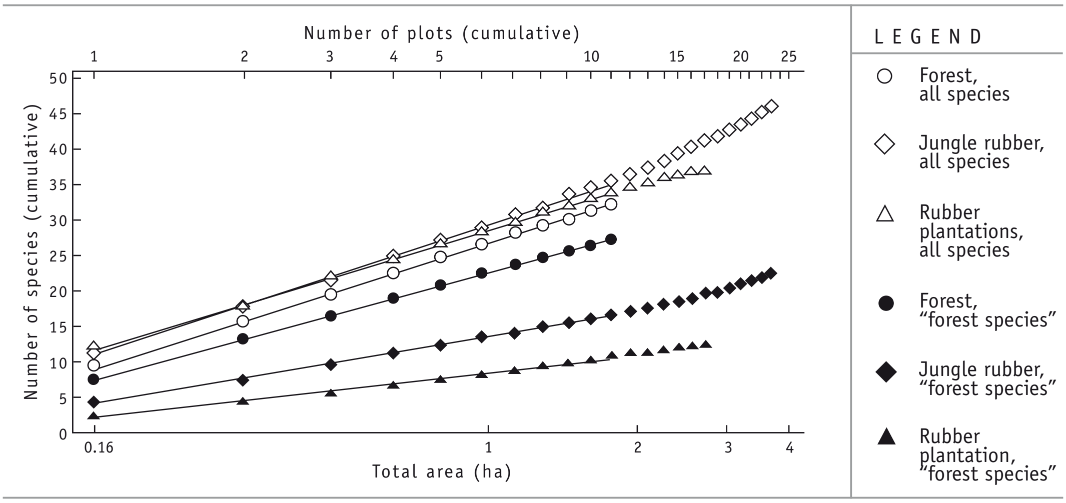
→Rubber slab containing a high percentage (about 45 percent) of dry rubber content.

→Micro-hydropower as non-financial reward for Lubuk Beringin village for conserving biodiverse jungle rubber systems.

Figure 15

Species-accumulation curves for the bird data of Danielsen and Heegaard, 1995.

Open symbols: all birds identified to species level. Filled symbols: subset of 'forest species' classified in habitat group 1: species mostly associated with the primary and old secondary forest interior.



Adapted from Figure 8 in Beukema *et al.*, 2007: 228

rubber agroforests (total of about 2 500 ha) if incentives are provided. The incentives local people requested include support to establish micro-hydro power plants, setting up of rubber nurseries and demonstration plots of improved rubber agroforests, and clonal plants of high yielding rubber trees for intensively managed rubber gardens elsewhere. Conservation agreements were signed by these four villages in 2006 (ICRAF, n.d.; Leimona and Joshi, 2010). The incentives provided then were seen only as an interim reward while a more permanent reward mechanism is being sought. RUPES is currently considering an eco-certification scheme for these complex rubber agroforests that will fetch a price premium for the natural rubber from the 'jungle' to be used in niche markets, such as 'green cars' and bicycle tyres. There is also a possibility of bundling biodiversity services together with other services, such as carbon or water quality (Leimona and Joshi, 2010).

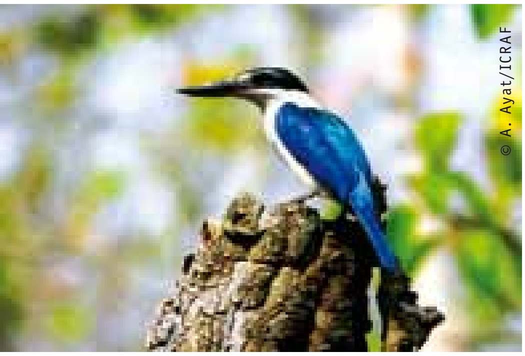
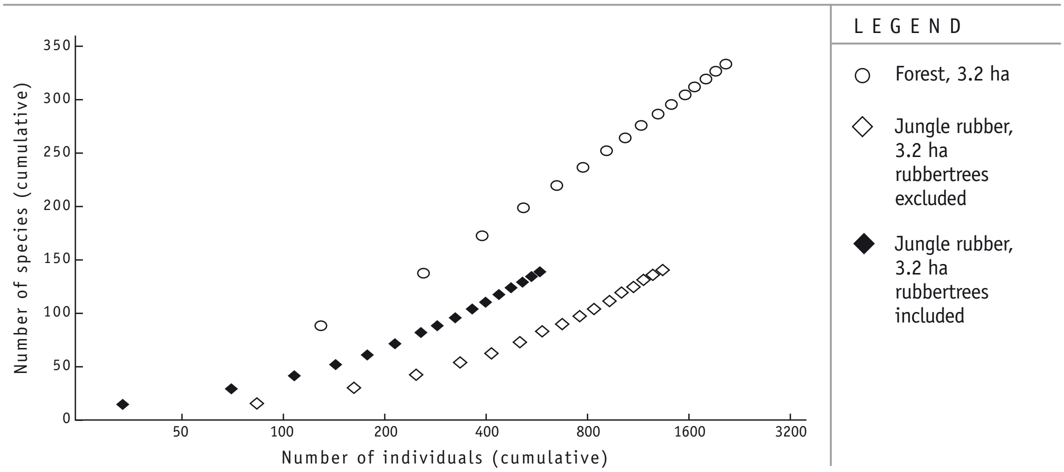


Figure 16

Species-accumulation curves for terrestrial pteridophytes in forests (dots), jungle rubber (diamonds) and rubber plantations (triangles).

Open symbols: all terrestrial pteridophyte species; filled symbols: 'forest species' subset. Plots were 0.16 ha each, non-adjacent and spread over a large area in Jambi province.



Adapted from Figure 7 in Beukema *et al.*, 2007: 228

The Bungo case study is a clear example on how biodiversity assessments are comprised of multiple layers of information. In this case, the generic relationship between rubber agroforestry and biodiversity has to be decomposed in at least four different levels, distinguishing between (a) plant and (b) animal levels of biodiversity, while considering biodiversity conservation at both the (c) plot and (d) landscape levels. Moreover, jungle rubber gardens also show the crucial relationship between biodiversity and land management over time because not only different management regimes influenced the recorded biodiversity level, but under the same management regime jungle rubber gardens of different ages host different levels of biodiversity.

Examples of animal biodiversity found in the forest and forest-edge habitat of Bungo district, where jungle rubber gardens often constitute a corridor between remaining forest patches (from left to right):

→ Collared kingfisher (*Halcyon chloris*).

→ Painted bronzeback snake (*Dendrelaphis pictus*).

→ Crab-eating macaque (*Macaca fascicularis*).

→ Indian momtjac (*Muntiacus muntiak*).

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