PAYMENTS *for* ECOSYSTEM SERVICES *and* FOOD SECURITY

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Editorial Note

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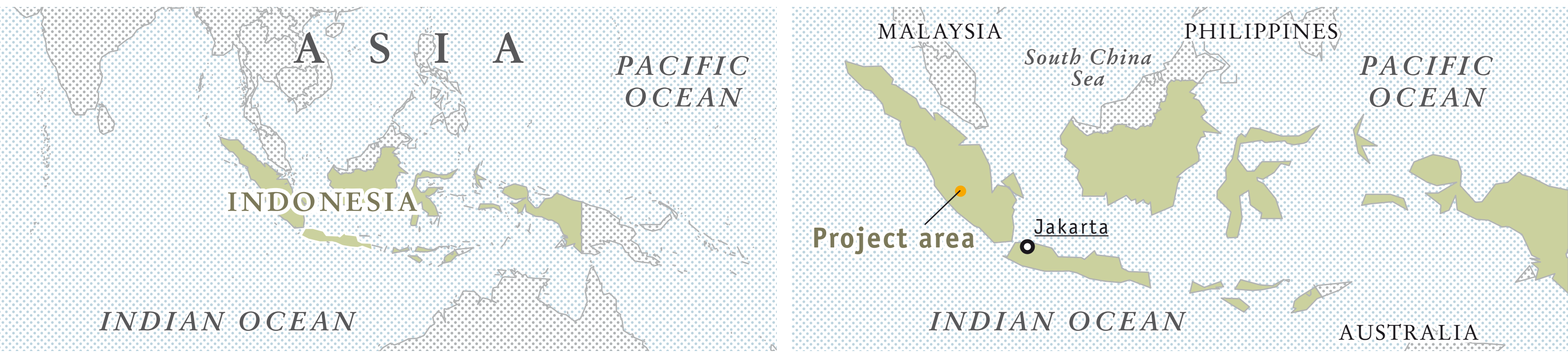
RUBBER AGROFORESTRY AND PES FOR PRESERVATION OF BIODIVERSITY IN BUNGO DISTRICT, SUMATRA

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

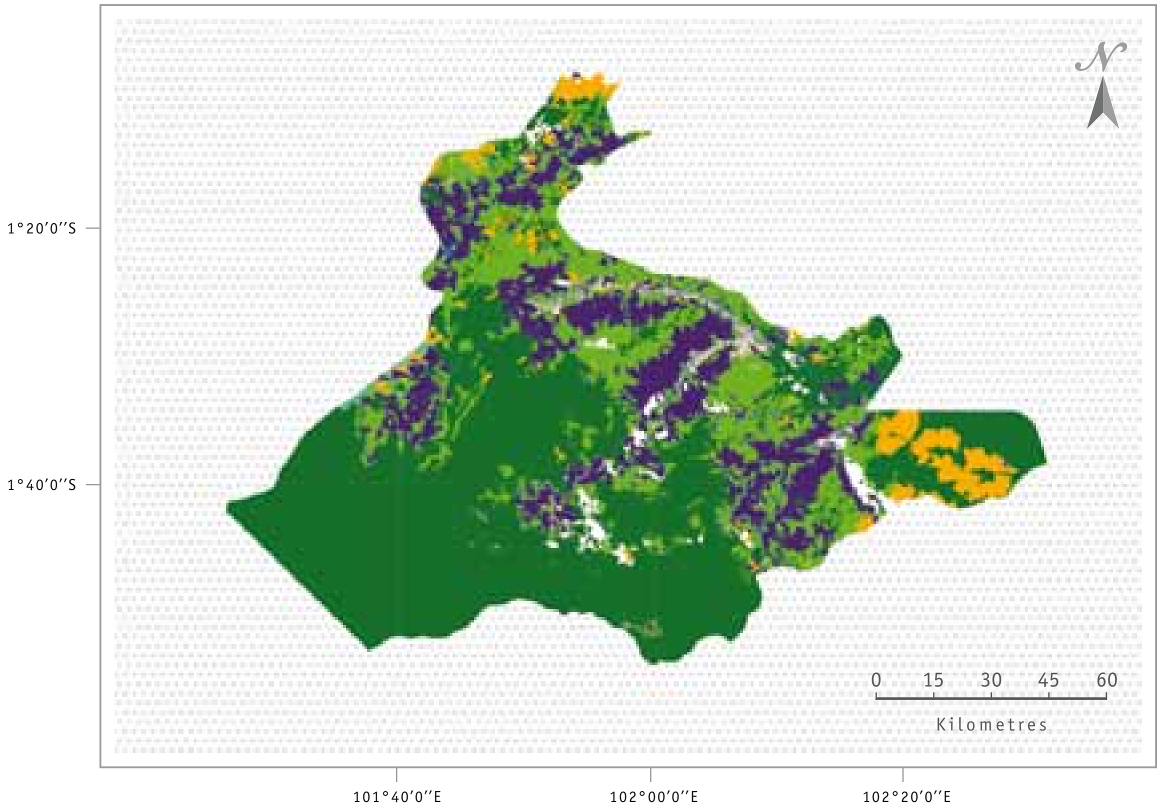


The introduction of the rubber tree (*Hevea brasiliensis*), naturally found in the floodplains forests along the Amazon River, began in Indonesia in the second half of the 19th century. In Sumatra and Borneo, rubber cultivation, initially restricted along rivers with good accessibility, rapidly spread 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 The district has been severely deforested (60 percent forest loss) and forests have been replaced by rubber and oil palm plantations, as well as other agricultural land uses.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):

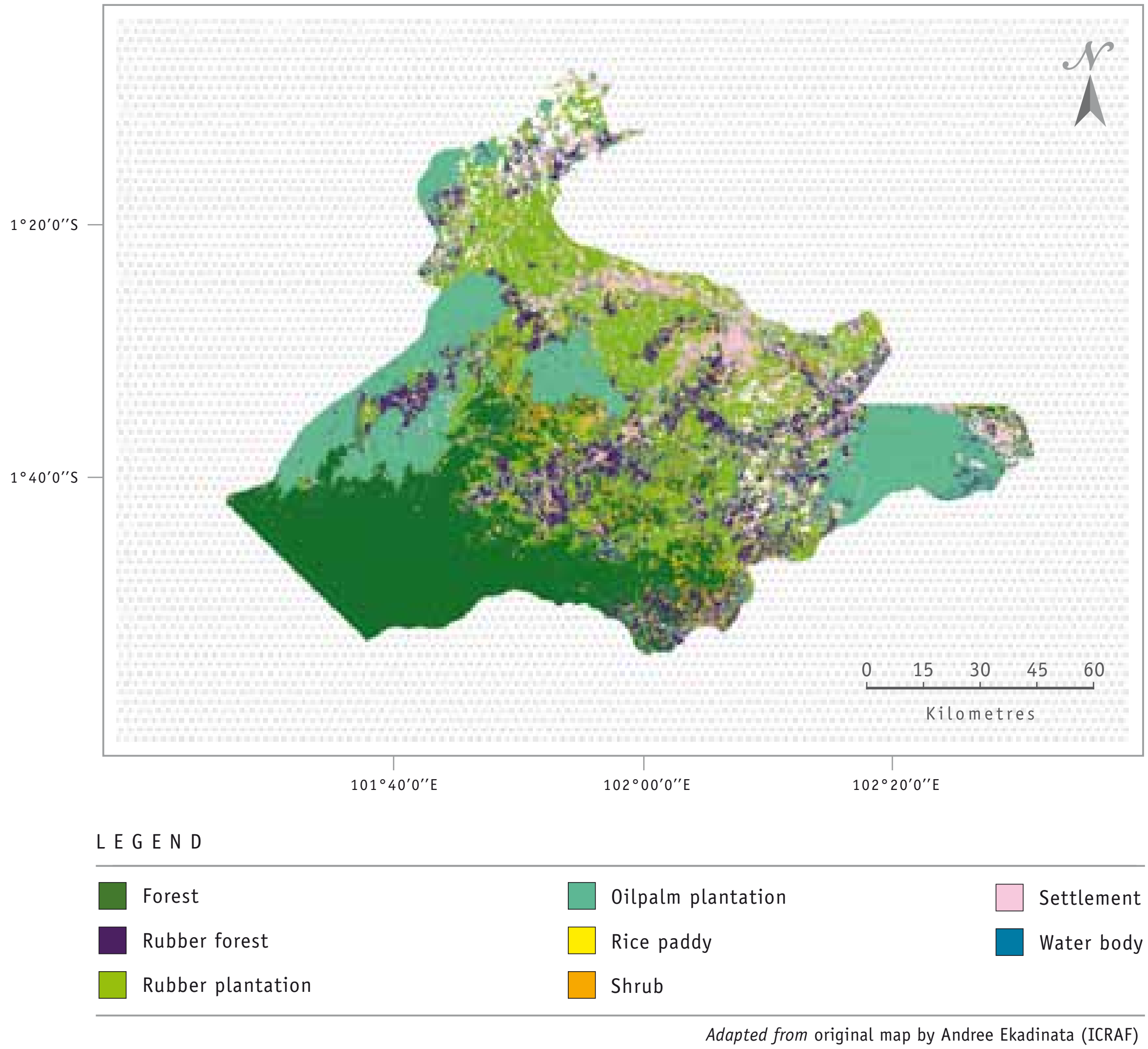
→ Surroundings of Lubuk Beringin, the first village granted with the legal right (*hutan desa*) by the Indonesian Government to manage state forests for their own prosperity.

→ 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



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.*,

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.

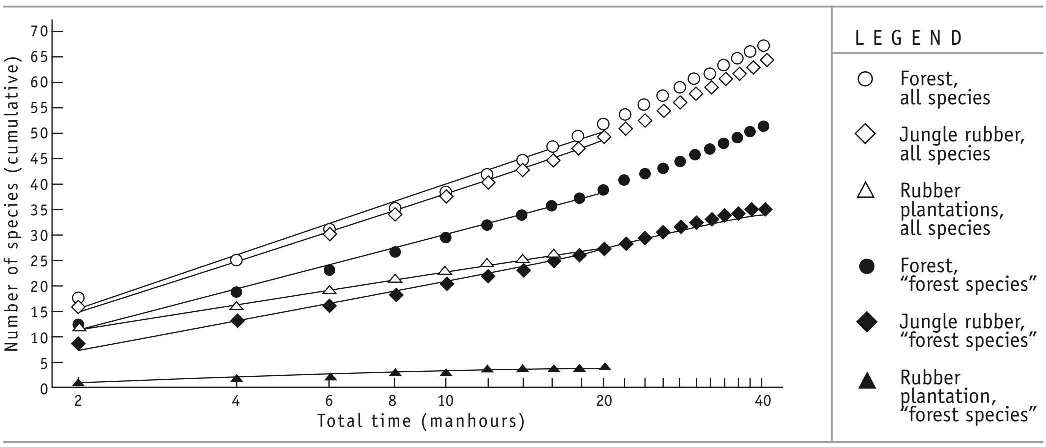
The same was true for terrestrial pteridophytes (Figure 16): for a subset of forest species, the number of species found was much lower in jungle rubber than in primary forest (Beukema *et al*. 2007).

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.

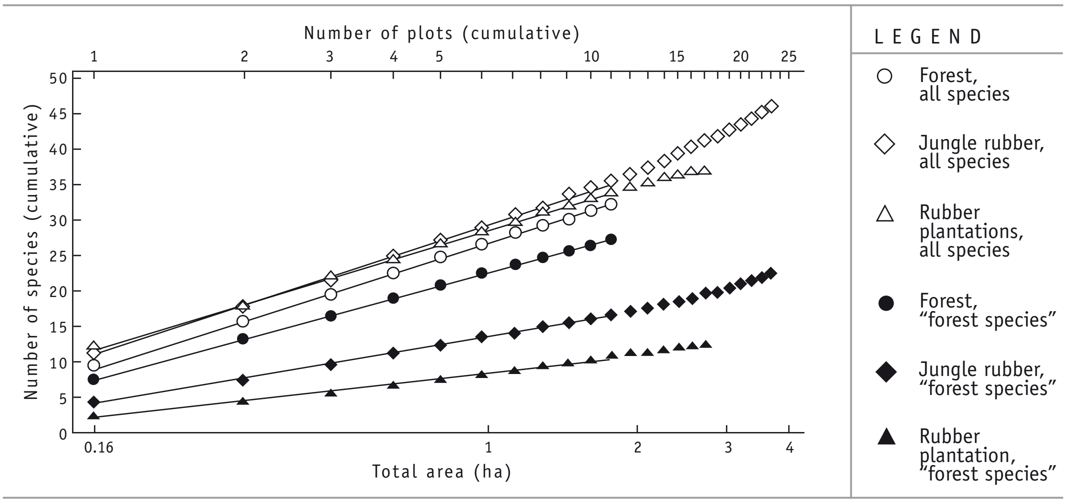
→ Rubber slab containing a high percentage (about 45 percent) of dry rubber content.

→ 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).

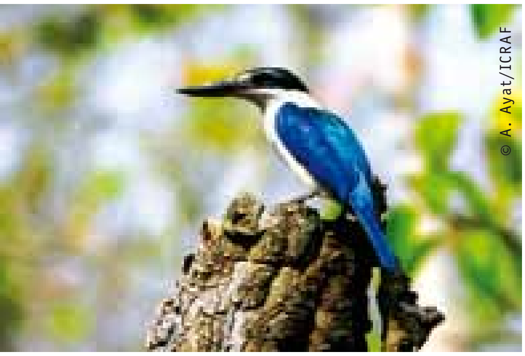
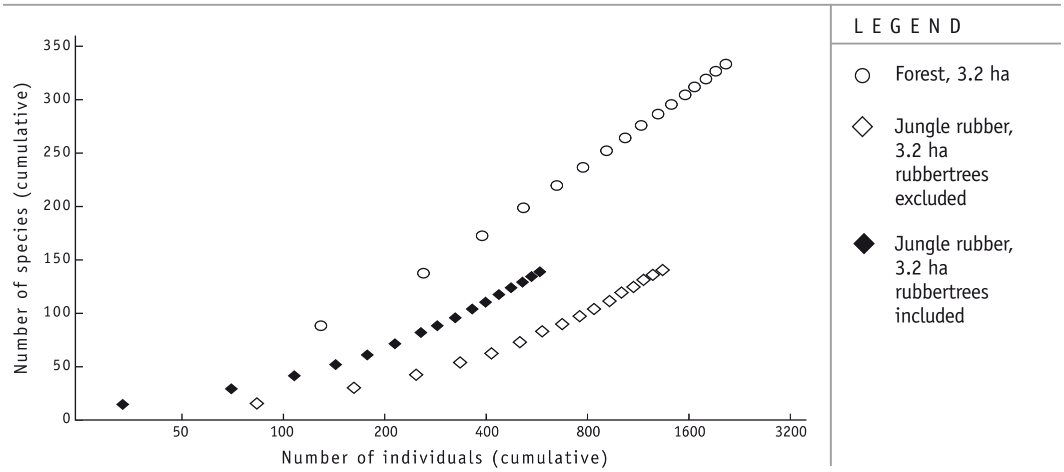


Figure 16

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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CHAPTER

4

COST-EFFECTIVE TARGETING OF PES

Katia Karousakis and Quiller Brooke
Organisation for Economic Co-operation and Development (OECD), Paris, France

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CONTENTS

ABSTRACT

Individuals or communities with the potential to influence the supply of ecosystem services will often differ in the magnitude of benefits they can provide, the risk that these services will otherwise be lost or the extent to which their management activities can enhance biodiversity and ecosystems, as well as the costs of service provision. This chapter discusses how PES programmes can be designed to address these issues and presents the tools and methods through which payments can be targeted to increase PES cost-effectiveness.

How payments for biodiversity and ecosystem services are targeted is critical in determining the cost-effectiveness of a PES programme. In most cases, the available budget for biodiversity and associated ecosystem services will be limited and competing with different demands. Cost-effective targeting of payments enables greater total benefits to be achieved with a given PES budget and can therefore also contribute to the long-term success of the programme.

Many PES programmes allocate uniform payments on a per hectare basis. This is cost effective if ecosystem service benefits and the costs of their provision are constant across space. In many cases however, this is unlikely. The more heterogeneous the costs and benefits are, the greater the cost-effectiveness gains that can be realized via targeted and differentiated payments. Indeed, more and more PES programmes are incorporating design elements to address this. This chapter examines the methods and tools that are available to target spatial heterogeneity in biodiversity and ecosystem service benefits, the threat of loss and the costs of their provision.

TARGETING ECOSYSTEM SERVICES WITH HIGH BENEFITS

The inherent complexity of biodiversity requires trade-offs between measurement accuracy and the cost of biodiversity assessments

Identifying areas with high biodiversity and ecosystem service benefits requires metrics and indicators to quantify them. Selecting an appropriate metric or indicator for PES that aims to enhance biodiversity conservation and sustainable use is not necessarily straightforward however. Unlike carbon, for example, which is measured in tonnes of carbon dioxide equivalents (tCO_2e), there is no single standardised metric to quantify biodiversity. The multidimensionality and the inherent complexity of biodiversity require trade-offs between the accuracy of a metric and the costs of development. The appropriate biodiversity metric or indicator selected for a PES programme may also depend on the specific objectives of the programme. Indeed, methodologies for constructing metrics and indicators tend to be tailored to specific local, regional and national programmes and their objectives. Examples of metrics and indicators used across two biodiversity PES programmes, namely the Victorian BushTender programme in Australia and the PES scheme implemented in the Assiniboine River watershed of east-central Saskatchewan province in Canada are presented in Box 1.

Box 1

Metrics and indicators used to target biodiversity benefits in the Victorian BushTender and a Canadian pilot PES

The Habitat Hectare methodology in the Victorian BushTender programme

The aim of Victorian BushTender programme in Australia is to improve the management of native vegetation on private land. To quantify biodiversity benefits, the BushTender programme uses the Habitat Hectare (HH) methodology. The HH is comprised of an assessment of the local benefits via the Biodiversity Benefits Index (BBI). The BBI is based on the proposed management practices; the conservation significance in terms of regional priorities through the Biodiversity Significance Score (BSS), the cost of conserving the land (*b*) and the size of the proposed land (ha). Potential plots are compared through an inverse auction, where landholders submit bids including information on the proposed area, the BBI and the required payment. The BSS is calculated separately to improve competition (DSE, 2009).

HH = BBI x ha
BBI = (BSS x HSS) *b*
where
HH = Habitat Hectare;
BBI = Biodiversity Benefits Index;
ha = area in hectares;
BSS = Biodiversity Significance Score;
HSS = Habitat Service Score; *b* = cost of bid

Targeting Waterfowl in a Canadian pilot PES programme

In Canada, a pilot PES programme initiated in 2008 to restore drained wetlands was undertaken in the Assiniboine River watershed of east-central Saskatchewan. The Environmental Benefits Index (EBI) was based on the incremental increase in predicted hatched waterfowl nests relative to the bid price. The EBI was based on the Ducks Unlimited Canada Waterfowl Productivity Model (DUC) which evaluated the potential of wetland restoration on each plot to increase the number of hatched waterfowl nests in the Assiniboine watershed. The EBI was based on wetland area restored, waterfowl density, existing wetland density and the percentage of cropland in a 4x4 mile block around the plot (Hill *et al.*, 2011).

The use of such metrics to better target ecosystem service payments can substantially enhance PES cost-effectiveness. In the Tasmanian Forest Conservation Fund programme, for example, a comparison of using the AUD/CVI[1] metric with a simpler AUD/ha[2] metric indicated an 18.6 percent gain in conservation outcomes. Comparing the additional conservation gains (valued at approximately AUD 3.3 million) with the costs of achieving those benefits (AUD 0.5 million), illustrate that the ratio of benefits to costs from investing in the CVI is 6.9:1. Similarly, Wunscher *et al.* (2006) simulated different targeting approaches for the Costa Rican PES and estimated that a scenario selecting highest scoring sites with the given budget would have resulted in 14 percent higher benefits than the current system of selecting sites (see Case Study 5 "PES in Costa Rica").

SPATIAL MAPPING TOOLS

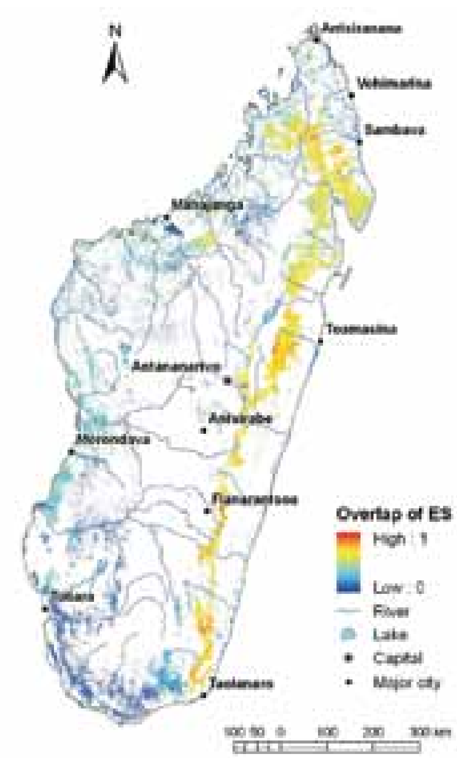
Spatial mapping tools are increasingly being used to discern the spatial heterogeneity in ecosystem costs and benefits. Several of these tools are emerging to help design PES systems at the regional and national level; however, there are increasingly initiatives of spatial mapping tools that are being developed at the international scale, including the UNEP-WCMC Carbon and Biodiversity Demonstration Atlas, ARtificial Intelligence for Ecosystem Services (ARIES),[3] the Integrated Valuation of Ecosystem Services and Trade-offs (InVEST)[4] and SENSOR.

Spatial mapping tools are increasingly being used to discern the spatial heterogeneity in ecosystem costs and benefits

To target ecosystem service payments in Madagascar, Wendland *et al.* (2010) examined the spatial distribution of biodiversity (proxied by vector data on species ranges of mammals, birds and amphibians), carbon and water quality. The left panel of Figure 17 depicts the degree of overlap between these three ecosystem services. The right panel further incorporates information on the probability of deforestation and the opportunity cost of the land to identify where payments could be most cost-effectively targeted. One example of a spatial mapping tool developed at the international level is the Carbon and Biodiversity Demonstration Atlas, produced by the UNEP World Conservation Monitoring Centre (UNEP-WCMC) (Kapos *et al.*, 2008). The Atlas includes regional maps as well as national maps for six tropical countries showing where areas of high biodiversity importance coincide with areas of high carbon storage. Figure 18 illustrates the national map for Panama, indicating that 20 percent of carbon is stored in high carbon, high biodiversity areas.

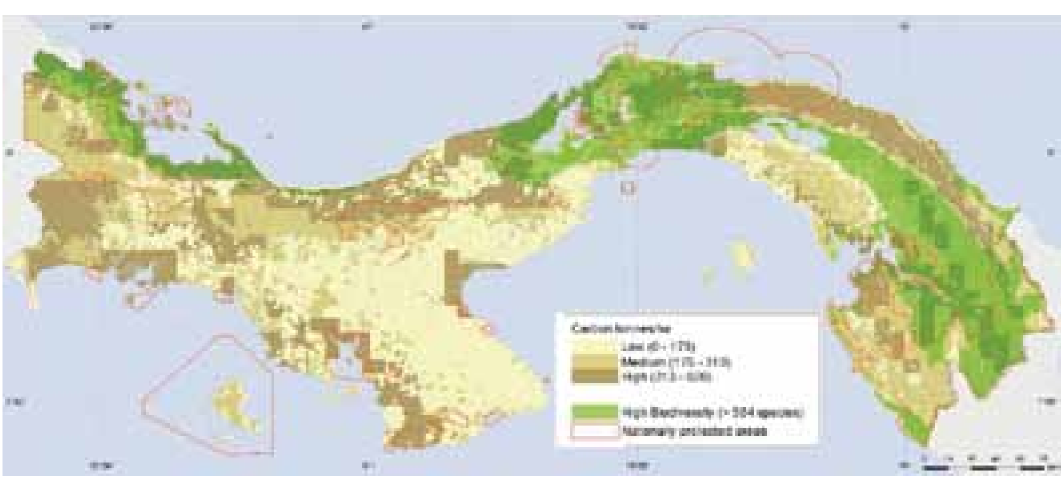
1 AUD/CVI: ratio of Australian Dollars (AUD) to the Conservation Values Index (CVI)
2 AUD/ha: ratio of Australian Dollars (AUD) per hectare of land
3 http://esd.uvm.edu/
4 http://www.naturalcapitalproject.org/

Figure 17
Targeting PES in Madagascar



Source: OECD, 2010

Figure 18
Example of a UNEP-WCMC national map: Panama



Source: OECD, 2010

To identify areas of high biodiversity importance for the regional maps, UNEP-WCMC uses six indicators for biodiversity, namely Conservation Internationals' Hotspots, WWF's Global 200 ecoregions, Birdlife International's Endemic Bird Areas, Amphibian Diversity Areas, Centers of Plant Diversity and the Alliance for Zero Extinction Sites. Areas of high biodiversity, as determined by UNEP-WCMC, are areas where at least four of the above-listed biodiversity conservation priority areas overlap, with areas in dark green indicating a greater degree of overlap.

The maps identify the different areas with high biodiversity importance. The maps do not necessarily identify areas with high biodiversity benefits in economic terms. Ideally, spatial maps on biodiversity benefits would incorporate the total economic value of these sites, with an assessment of both direct and indirect use values.

A number of spatial mapping initiatives are currently underway and are in different stages of development. These include ARtificial Intelligence for Ecosystem Services (ARIES) (Villa *et al.*, 2009); InVest (Tallis *et al.*, 2010); the United States Geological Survey (USGS) Global Ecosystems initiative;[5] and SENSOR (Sustainability Impact Assessment: Tools for Environmental, Social and Economic Effects of Multifunctional Land Use in European Regions).[6]

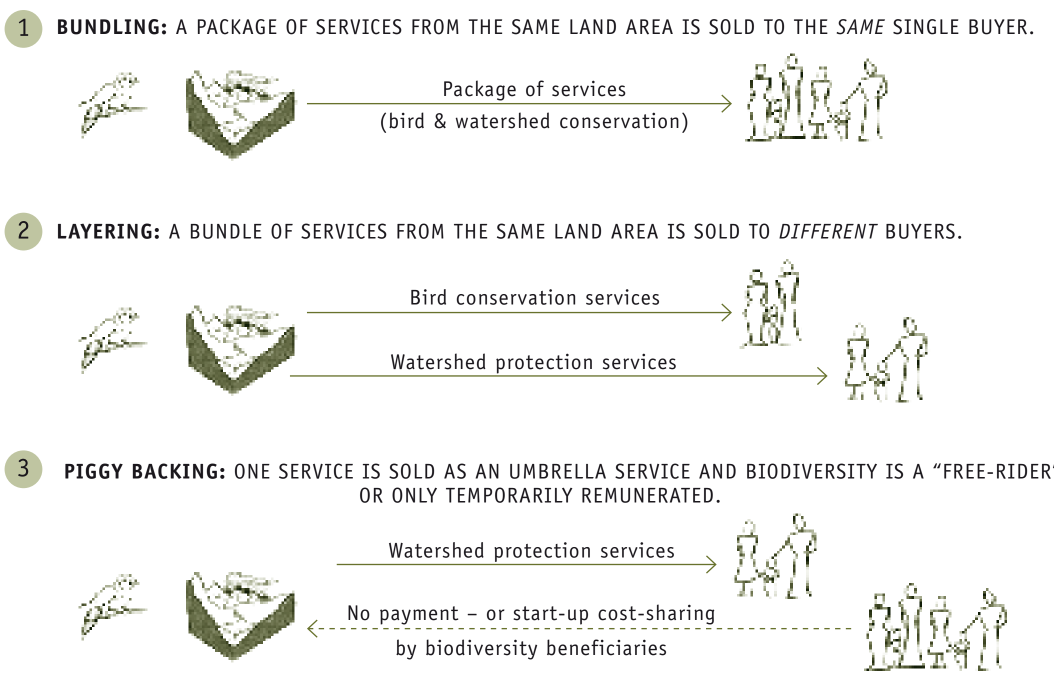
PES objectives must be clear, potential trade-offs recognised and safeguards developed to prevent adverse collateral impacts

As suggested in the Madagascar example above (Figure 17), PES programmes can simultaneously target multiple ecosystem service benefits. Bundling or layering (Figure 19) can allow a broader range of ecosystem service benefits to be obtained in a cost-effective manner, avoiding the need for multiple programmes, reducing transaction costs and programme overlap. Multiple ecosystem service provisions can help ensure that all aspects of an ecosystem on enrolled land are properly managed, increasing the asset value of the ecosystem. PES targeting multiple ecosystem services can enable the landholder to maximise potential payments received, such that conservation becomes more economically feasible, enabling greater ecosystem service provision.

The feasibility of targeting multiple ecosystem services simultaneously depends on the degree of spatial correlation between different types of ecosystem services. Spatial mapping tools help to identify where multiple service benefits coincide. Though there may often be synergies in service provision (e.g. avoided deforestation results in both biodiversity and carbon benefits), there are cases when trade-offs can also arise (Nelson *et al.*, 2008). For example, whereas native and mixed crops provide biodiversity benefits, monocultures of fast-growing tree species such as *Eucalyptus* may provide more rapid carbon sequestration benefits. Farley *et al.*

5 http://rmgsc.cr.usgs.gov/ecosystems/
6 http://www.ip-sensor.org

Figure 19
Marketing biodiversity joint service provision

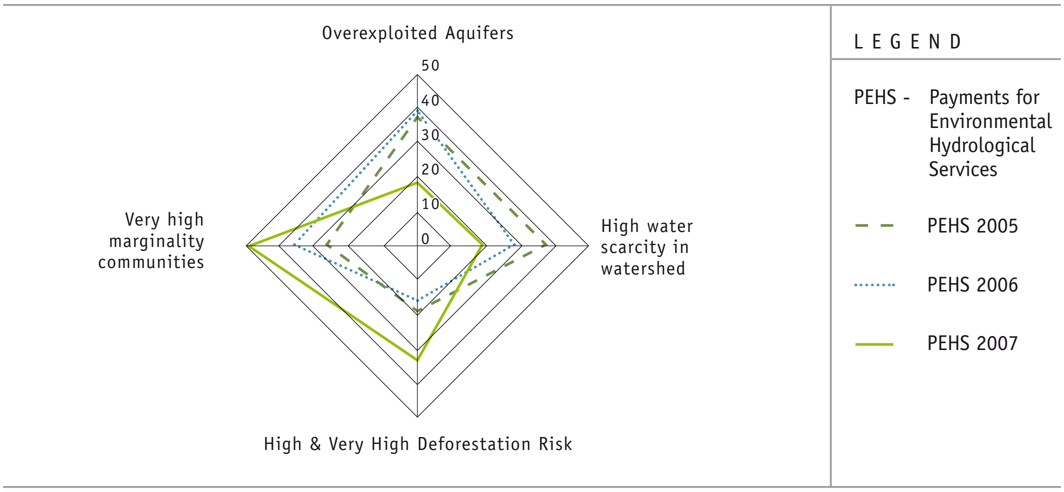


Source: OECD, 2010

(2005) highlighted this problem in West Africa, where carbon sequestration (i.e. afforestation/ reforestation) projects can negatively affect water regimes and biodiversity. The ultimate objective of the PES programme must therefore be clear, potential trade-offs recognised and safeguards may be needed to prevent adverse impacts on other ecosystem services (Karousakis, 2009). In this context, environmental benefit indices and scoring approaches become not only a way of evaluating the quality of potential contract benefits, but are also mechanisms through which discrete ecosystem service priorities are traded off against each other. Any weights associated with an Environmental Benefits Index (EBI) or scoring mechanism can also be modified in sequential PES sign-up rounds to reconcile trade-offs. This has been done, for example, in the Mexican PEHS[7] programme (Figure 20) where weights have been adjusted over time to better address the policy priorities. Similar targeting methods have been used to allocate payments in the Socio Bosque programme in Ecuador. Based on a system of scores,

7 Payments for Environmental Hydrological Services (*Pago de Services Ambientales Hydrologicas - Mexico*)

Figure 20
Targeting PEHS in Mexico



Source: OECD, 2010

land area has been classified into three categories of priority: priority 1 (scoring from 12.1 to 25); priority 2 (7.1 to 12) and priority 3 (0 to 7). The scores are based on high deforestation pressure, storage of carbon in biomass, water supply and poverty alleviation.

Though these types of targeting approaches entail higher transaction costs, experience with their use suggests that the resulting cost-effectiveness gains are improved. There are also other types of PES design characteristics that can be introduced into the programme to reduce transaction costs. In the Costa Rican PES, for example, private forest landholders are required to have a minimum of one hectare to receive payments for reforestation and two hectares in the case of forest protection. The maximum area for which payments can be received is 300 hectares (and 600 hectares for indigenous peoples' reserves) (Grieg-Gran *et al.*, 2005). Aggregating small projects is also possible to help reduce the transaction costs associated with a payment contract. These types of PES design elements can help to ensure more equitable participation in the PES programme and help to reduce administrative costs.

TARGETING ECOSYSTEMS SERVICES AT RISK OF LOSS OR DEGRADATION

In addition to targeting payments to ecosystem services with the highest benefits, it is essential to ensure that any payment leads to additional benefits relative to the business-as-usual scenario. For example, payments for habitat protection are only additional if in their absence the habitat would be degraded or lost. Information on the business-as-usual or baseline scenario is critical in ensuring PES additionality. Clear understanding of whether or not ecosystem services are at risk of loss or degradation is therefore needed. Historical and current trend data on biodiversity and ecosystem service loss are a starting point and are needed to develop future reference projections. Though this can be a complex task, there are different ways this can be undertaken. For example, to target PES in Madagascar, Wendland *et al*. (2010) estimate the probability of deforestation (via a multivariate probit model) by examining distance to roads and footpaths, elevation, slope, population density, mean annual per capita expenditure and other characteristics. A similar approach is used to assess deforestation risk in the Mexican PEHS programme. In this case, the variables used to estimate deforestation risk include distance to the nearest town and city, slope, whether it is an agricultural frontier and if it is located in a natural protected area.

Information on the baseline scenario is critical to ensuring the additionality of PES projects

TARGETING PROVIDERS WITH LOW OPPORTUNITY COSTS

Finally, PES programmes can increase their cost-effectiveness if, given sites with identical ecosystem service benefits and risk of degradation or loss, payments are differentiated and prioritised to those sites where landholders have lower opportunity costs of alternative land uses. In the Costa Rican PES, for example, Wunscher *et al.* (2006) illustrate that differentiating payments according to opportunity costs could allow the enrolment of almost twice the area of land, representing more than double the environmental benefits per cost (see Case Study 5 "PES in Costa Rica").

Obtaining accurate information on ecosystem providers' opportunity costs is not straightforward as they have an incentive to overstate these costs in an effort to extract information rents via higher payments. Programme administrators have a number of options to assist revelation of the landholder's true opportunity costs. Specifically, they can gather additional information in the form of costly-to-fake signals or they can use inverse auctions.[8]

Information on ecosystem supplier attributes and activities which are correlated with their opportunity costs can be used to infer the correct price. The information should be based on costly-to-fake signals, for example, distance to markets, current land use, assessed value, or labour and production inputs. Readily available market information can also be used and incorporated into a model to estimate opportunity costs. In the USA Conservation Reserve

8 Screening contracts can be used in theory, but this is complicated in practice; see Ferraro (2008)

Program, for example, local land rental rates are combined with information on field soil types, a proxy for productivity, to give a reasonable indication of the opportunity costs of retiring agricultural land. This is then used as a maximum acceptable price, removing the landholders' ability to claim unreasonably high payments. To proxy for opportunity costs in Madagascar, Wendland *et al.* (2010) use data on the opportunity costs of agriculture and livestock produced by Naidoo and Iwamura (2007). Naidoo and Iwamura compiled information on crop productivity and distribution for 42 crop types, livestock density and estimates of meat produced from a carcass and producer prices to measure the gross economic rents of agricultural land across the globe. Wendland *et al.* (2010) clipped this global data to Madagascar's boundaries. Gross economic rents ranged from USD 0 to 529 per hectare for Madagascar, with a mean value of USD 45 per ha, per year. The value of USD 91 per ha, per year (one standard deviation) was used as the cut-off to exclude areas of high opportunity costs.

However, obtaining information on costly-to-fake signals still incurs research costs. The efficiency of the payment will directly depend on the quality of this research and the strength of the correlation between the signal and the opportunity costs, which must be assessed on a case-by-case basis.

Exploiting competition between ecosystem service suppliers for conservation contracts through inverse auctions can provide an effective cost-revelation mechanism. Where suppliers are heterogeneous in their opportunity costs and demand for contracts exceeds supply (i.e. the conservation budget), competitive procurement auctions are possible.

The recognition of the potential gains from the use of inverse auctions as a payment allocation mechanism has stimulated heightened interest from policy-makers. Though their use in PES programmes is not yet common, they are becoming more widespread in developed and developing countries. Inverse auctions have been used to allocate PES contracts in Australia, Canada, Finland, Germany, Indonesia, Tanzania, the United Kingdom and the USA (Claassen, 2009; DSE, 2009; EAMCEF, 2007; Hill *et al.*, 2011; Jack, 2009; Juutinen and Ollikainen, 2010; Latacz-Lohmann and Schilizzi, 2005).

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