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# Spatial targeting of payments for environmental services: A tool for boosting conservation benefits

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## ABSTRACT

Payments for environmental services (PES) have become an increasingly popular market-based instrument to translate external, non-market environmental services into financial incentives for landowners to preserve the ecosystems that provide the services. However, lack of spatial differentiation in the targeting mechanism may lead to efficiency losses. Addressing this challenge, we construct an applied site selection tool, which takes into account three variables that vary in space: environmental services provided, risks of losing those services, and participation costs. Using data from Costa Rica's Nicoya Peninsula, we empirically test the tool's potential to increase the financial efficiency of the forest-focused PES program in place. Results show that, given a fixed budget, efficiency increases radically if per hectare payments are aligned to landowners' heterogeneity in participation costs, involving opportunity, transaction and direct costs of protection, respectively. Selecting sites based on environmental service potential also moderately increases efficiency. Overall additionality could in the best case be doubled, but remains generally limited due to current low deforestation risks prevailing in Costa Rica. To take advantage of the high efficiency potentials of flexible payments, we propose inverse auction systems as a cost-effective approach for the determination of micro-level participation costs.

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## 1. Introduction

Payments for environmental services (PES) have become an increasingly popular market-based instrument to translate external, non-market environmental services into financial incentives for landowners to preserve the ecosystems that provide the services. We follow Wunder (2005) in defining PES as a voluntary transaction where a well-defined environmental service (or a land-use likely to secure that service) is being bought by a minimum of one buyer from a minimum of one provider if and only if the provider secures environmental service provision. However, the self-selecting nature of volun-

tary participation in PES programs may result in contract allocations that add relatively little to service provision levels. When the number of applications to participate in the PES program exceeds available financing, the service buyer can select among applicant sites to maximize the program's financial efficiency. We argue that conservation service buyers should ideally distinguish the spatially variable level of (i) environmental service provision (benefits), (ii) risk of environmental service loss (e.g. through deforestation), and (iii) landowner's participation cost (sum of opportunity, transaction, and protection costs, which jointly determine the minimum payment required for the landowner to participate).

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While benefits and costs are widely acknowledged targeting criteria (e.g. Babcock et al., 1997; Ferraro, 2004; Barton et al., 2003; Claassen et al., 2008-this issue), they are not fully considered in most current PES schemes. Moreover, the risk of service loss, here due to deforestation, deserves additional attention. As Hartshorn et al. (2005, p.12) state, “paying for forest protection on land that requires no protective measures is an inefficient use of scarce conservation funds”. Wunder (2005) argues that the future of PES largely depends on the programs’ ability to demonstrate clear additionality. Yet, additionality is only provided if environmental services are secured that otherwise would have been lost. Preliminary evidence on Costa Rica’s nationwide PSA<sup>1</sup> program, which is the focus of our study, suggests that the program’s additionality is low.<sup>2</sup> Additionality is not a selection criterion in the PSA program; it would pay all plots that provide environmental services if financial resources were available, irrespective of additionality (Pagiola, 2008-this issue). Accordingly, Ortiz et al. (2003) find that 76.8% of the forest area under PSA contracts would likely have been conserved or managed with limited forest interventions in the absence of PSA. Hartshorn et al. (2005) also find more than 70% of PSA forest protection contracts to be on land with production capacities that only allow forest management/protection (51%), or severely limited agriculture (20%).<sup>3</sup> Pfaff et al. (unpublished) find that annually only 0.08% of the PSA contracted forest would have been cleared in the absence of payments.<sup>4</sup> Sierra and Russman (2006) conclude that PSA has limited immediate effects on forest conservation on the Osa Peninsula. Finally, at the micro-scale Sills et al. (2006) detected statistically significant but very small effects on land-use patterns between PSA and non-PSA land.

Costa Rica’s PSA program is implemented by the National Fund for Forest Financing (FONAFIFO). It currently recognizes four different activities as delivering forest environmental services: (i) forest protection, (ii) timber-plantation establishment, (iii) natural forest regeneration, and (iv) agroforestry system establishment. Between 1997 and 2005, most funds (80.7%) were spent on forest protection which, because of its dominance in the PSA program, is the focus of this study. Forest protection payments (approx. US\$40/ha/year until 2005, and US\$64/ha/year since 2006) are identical across the entire country and attract applications for a land area much larger than the program budget can pay for. For example, in 2006, FONAFIFO’s regional office in Nicoya received applications for about 12,000 hectares (ha), but only had funds available for 2000 ha (J.A. Jiménez Fajardo, pers. comm., 2007).<sup>5</sup> PSA implementers currently base the selection of plots primarily

on predefined priority areas. Site differentiation, therefore, is limited to “inside” versus “outside” these areas. Although the program aims to generate carbon, biodiversity, hydrologic and scenic beauty services, no attempt is made to quantify the delivery of these services. In summary, it appears that the Costa Rican scheme could benefit significantly from improved spatial targeting.

Babcock et al. (1997) classify targeting approaches for conservation programs into those that target (i) benefits, (ii) costs, or (iii) benefit-to-cost ratios. Examples of benefit targeting approaches are Powell et al. (2000) and Rodrigues et al. (2003), conducting gap analyses to identify high-benefit priority areas for biodiversity conservation, and Imbach (2005), targeting multiple environmental service objectives and deforestation probabilities. Chomitz et al. (2006) is an example of cost targeting where negative correlation between costs and biodiversity leads to low-cost/high-benefit solutions. Examples of approaches targeting benefit-to-cost ratios are presented in Ferraro (2003) and Barton et al. (2003). There are also approaches that target benefits and costs but not their ratio, e.g. linear scoring functions that include costs as an element of the scoring equation (Claassen et al., 2008-this issue) and non-parametric, multi-objective approaches like distance function rankings (Ferraro, 2004).

We contribute to the targeting literature by presenting a tool that targets benefit-to-cost ratios incorporating multiple objectives and explicitly considering the risk of environmental service loss as a spatial variable to determine benefit additionality. We build our approach on Imbach (2005), but employ different assumptions for environmental service distribution, partially different data (e.g. on deforestation probabilities) and, most important, integrate micro-level participation costs. Our approach comes closest to a recent study by Alix-Garcia et al. (2005) on Mexico, who also examine cost-benefit targeting under consideration of deforestation risks.

Using data from Costa Rica’s Nicoya Peninsula, we empirically test the potential of the targeting tool to boost the financial efficiency of the PSA program in terms of expected additional environmental services (benefits) per dollar spent. In addition, we also compare the results to those obtained from partial consideration of targeting criteria. We use simple targeting criteria with somewhat crude assumptions to illustrate the general usefulness of the tool. For real-world application, the data would need to be refined and complemented, for example by including available information on species to elaborate biodiversity service estimates.

The tool addresses three kinds of possible inefficiencies in a PES program, as described by Pagiola (in press): (i) the desired land-use activity would be adopted even in the absence of payments; (ii) the payment is not high enough to lead to the socially desirable land use; (iii) the payment encourages a land use with positive externalities that are worth less than the cost. We address (i) by applying deforestation probabilities: a forest at low risk of deforestation would provide less additionality if enrolled, so the tool gives it a lower likelihood of being selected. We address (ii) through flexible payments, aligned to landowner participation costs. While we cannot guarantee that the additional service value is larger than its provision cost, we reduce the likelihood of type (iii) inefficiency by selecting first the sites with the largest service-cost ratio. To fully address (iii) would

<sup>1</sup> PSA (Pagos por Servicios Ambientales) refers to Costa Rica’s PES program.

<sup>2</sup> For other examples of national PES programs which only partially consider spatial differences see Muñoz-Piña et al. (2008-this issue) on Mexico and Bennett (2008-this issue) on China.

<sup>3</sup> Note that Hartshorn et al. (2005) refer to the number of contracts while the more important unit would be area.

<sup>4</sup> We assume here that deforestation is the only land-use change factor affecting service provision. We discuss this restrictive assumption in Section 7.

<sup>5</sup> See Pagiola (2008-this issue) for a detailed description of the Costa Rican PSA program.

require a monetary valuation of site additionality. Because of the difficulty of transferring existing benefit estimates to new sites (e.g., Kirchhoff et al., 1997), we refrain from monetarization, and apply instead a scoring technique. However, type (iii) inefficiency would be avoided if PES was allocated directly by a service buyer with a defined marginal willingness to pay (expressed in dollars per score unit).

The remainder of the article proceeds as follows. In Section 2, we describe the general concepts behind the site-selection mechanism. We then present the study area and data (Section 3) and define the simulated targeting approaches (Section 4). The corresponding empirical results are presented in Section 5. In a sensitivity analysis (Section 6), we test the stability of results. Section 7 discusses the findings, while Section 8 gives overall conclusions.

## 2. Concept and definitions

The three principal variables (services, costs, and deforestation risk) are represented by six main spatial data sets. Four of them are used to estimate the levels of environmental services (hydrological services, biodiversity conservation, carbon sequestration, and landscape beauty services), while the others are used to estimate deforestation probabilities and participation costs.

Each of the four services is measured in service-specific units and so cannot be compared directly (see Section 3). Therefore, a z-normalization was applied, yielding comparable scores with a mean equal to zero and standard deviation and variance equal to one (Hogg and Craig, 1978).<sup>6</sup> The z-value normalization for data sets where higher values are given priority over lower values (e.g. slope) has the following formula:

$$z_i = \frac{x_i - \text{mean}(x)}{\text{S.D.}(x)} \quad (1)$$

with  $x_i$  being the observed value of the  $i^{\text{th}}$  pixel (30 meters). For data sets where lower values are given priority over higher values (e.g. distance of plots to existing or proposed protected areas) the z-normalization has the following formula:

$$z_i = \frac{\text{mean}(x) - x_i}{\text{S.D.}(x)} \quad (2)$$

Additionality ( $e$ ) is defined to be the product of environmental service provision ( $u$ ), which equals the sum of z-values for environmental services, and deforestation probability ( $r$ ):

$$e = u \cdot r \quad (3)$$

We assume that property owners are always willing to participate if the per hectare PES payment ( $C_{\text{payment}}$ ) exceeds the sum of their per hectare opportunity ( $C_{\text{opp}}$ ), protection ( $C_p$ ) and transaction costs ( $C_t$ ).<sup>7</sup> We define  $\gamma_i \in \{0, 1\}$  as an indicator variable reflecting participation:

$$\gamma_i = 1 \text{ if } C_{\text{opp}} + C_p + C_t < C_{\text{payment}}; \gamma_i = 0 \text{ otherwise} \quad (4)$$

Opportunity costs refer to the difference in income between the most profitable land use (before PES) and forest conservation. Protection costs relate to active forest-protection efforts (e.g. firebreaks, fencing off cattle). The landowners' transaction costs are all residual PES-related landowner expenses for contract establishment and maintenance (e.g. travel expenses, information gathering, and external monitoring). We define the sum of these three cost elements as the participation cost. If payment levels are flexible and aligned to participation costs, we can establish individual site-specific benefit–cost ratios.

## 3. Study area and data

The PSA program is well established on the Nicoya Peninsula in northwestern Costa Rica. During 2004 alone, FONAFIFO enrolled 181 plots with an area of 12,244 ha in the Tempisque Conservation Area, an administrative unit which consists mainly of the Nicoya Peninsula (FONAFIFO, 2004). Of these, 96.3% were enrolled for forest protection and the remaining 3.7% for the establishment of timber plantations.

A field survey with random sampling, interviewing 107 forest owners from a frame population of 4266 ranchers on Nicoya Peninsula, was used to estimate opportunity, transaction, and protection costs (Wünscher, in press). Levels of environmental services and deforestation risks were determined for the forests on these 107 sample sites, which contain 3736 ha of forest. Fig. 1 shows the location of the sample sites. Ideal estimation of environmental service delivery requires exceptionally complex and detailed data. We use simplifying assumptions to adapt estimates to demand and available data. Our service estimates are presented below. Alternative simplified approaches for Costa Rica can be found, for example, in Imbach (2005), Barton et al. (2003), and Tattenbach et al. (2006).

### 3.1. Biodiversity services

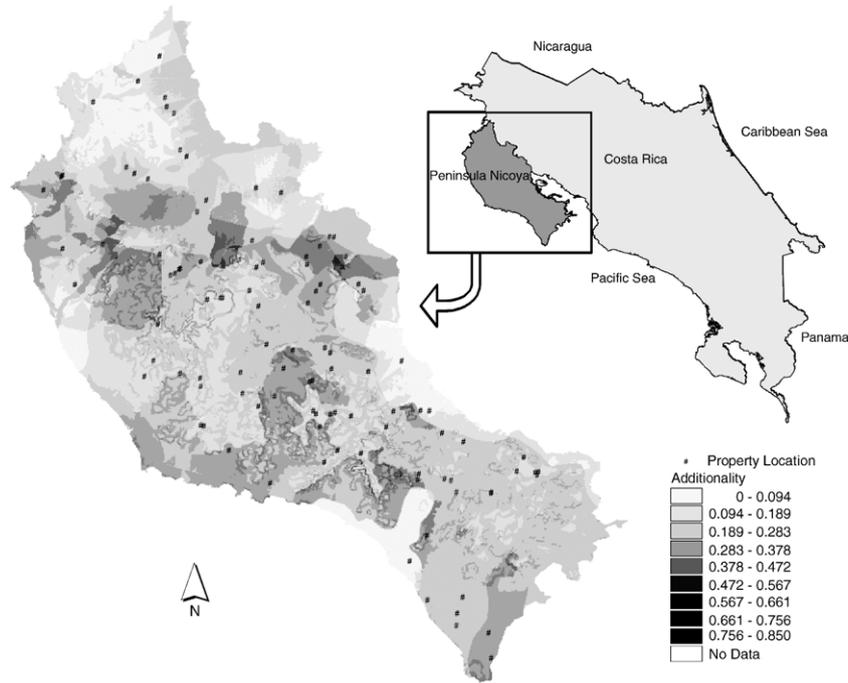
We define biodiversity services here as contributions towards the conservation of species communities in their natural forest habitat. Forest conversion to other land uses destroys natural habitat, causing the decline and, ultimately, extinction of forest-dwelling species. Forest biodiversity services would ideally be estimated considering, among others, species distribution, representation, minimum sustainable population size and area requirements<sup>8</sup> (e.g. Rodrigues et al., 2003; Sanderson et al., 2003; Faith et al., 2001; Powell et al., 2000).

Although biodiversity data has improved significantly in recent years (Rodrigues et al., 2003), adequate data is still unavailable for many parts of the world. In our simplified estimation approach, we thus use habitat types (life zones) from Holdridge (1967) as a surrogate for biodiversity. We measure their current representation in the protected areas of Nicoya

<sup>6</sup> See Ferraro (2004) for alternative standardization techniques.

<sup>7</sup> See Antle and Valdivia (2006) for a similar approach.

<sup>8</sup> Other factors include irreplaceability, endemism and risk of extinction, habitat quality, connectivity and protection status as well as present and future threats including environmental, demographic, and genetic stochasticity.



**Fig. 1 – Additionality and location of interviewed properties on Nicoya Peninsula. Source: own data.**

Peninsula, and set a target of protecting at least 20% of the area of each life zone. Thus, the least represented life zones received the highest conservation priority. The mean representation deficit within our sample sites is 12.8% (Table 1).<sup>9</sup> Furthermore, we use connectivity as a selection criterion. We measure the distance (in meters) to already existing protected areas and conservation corridors on private land that were proposed by the so-called GRUAS project (García, 1996). Plots close to these areas are given priority, in order to increase connectivity. The mean connectivity distance in the sample sites is 3005 m (Table 1). The z-values for representation and connectivity are combined to compute an aggregate mean z-value for total biodiversity–service provision, giving equal weight to the two sub-criteria.<sup>10</sup>

### 3.2. Carbon services

We define carbon services of forest conservation as the mitigation of carbon releases (carbon storage) through avoided deforestation, which is not recognized by the Kyoto framework but is currently traded on voluntary carbon markets. Ideal estimation of mitigation services would require information on the amount of stored carbon before and after land-use change and the risk of land-use change occurring. For the latter, we use the deforestation rate estimates by Pfaff and

Sanchez-Azofeifa (2004). Lacking data on site specific forest maturity, we assign each Holdridge lifezone a primary forest per hectare amount of stored carbon, which overestimates carbon storage in secondary forest. Life zone specific carbon storage quantities are derived from data on biomass per hectare (Imbach et al., 2005) assuming a carbon content of 50%

**Table 1 – Descriptive statistics of variables in the Nicoya Peninsula sample plots**

	Mean	Min.	Max.	S.D.
Representation deficit	12.8	0.0	19.3	4.9
Holdridge Lifezone (%)				
Distance to existing or proposed PA (m)	3005	0	9191	2154
Carbon storage (t)	126.2	94.0	135.8	9.4
Slope (%)	3.8	0	68.5	8.7
Water consumption (liters/hour/ha)	16.2	0.0	208.4	45.0
Visibility (number viewpoints)	9.0	0.0	30.0	7.4
Opportunity cost (US\$/ha/year)	56.09	-210.80	624.56	127.16
Protection cost (US\$/ha/year)	3.56	0.11	23.07	3.66
5 year deforestation probability (%)	3.86	2.09	6.81	1.24
Additionality (service scores/ha)	2.47	0.40	7.72	1.18
Score (service score/ha)	61.76	18.16	131.86	18.47
WaterScore (service score/ha)	7.080	0.003	79.736	11.753

Note: n=107.  
Source: Own computations.

<sup>9</sup> Due to limited computing capacity our approach is static and not iterative as, for example, the approach by Barton et al. (2003) who consider the change of targeting parameters with every selected parcel.

<sup>10</sup> Weights could also be distributed differently, but for lack of a clear weight preference we set them equal.

(IPCC, 1996). We assume that, if deforested, land-use converts to pasture with a biomass carbon storage of 5 tons of carbon per hectare (tC/ha)<sup>11</sup> (Rojas Molina, 2005). The crudely estimated mean carbon storage potential of our sites is 126.2 tC/ha (Table 1).

### 3.3. Hydrological services

We define hydrological services as the contribution of conserved forests to the natural supply of freshwater. Vegetation cover and soil management influence the interception, infiltration, storage, runoff, and evapotranspiration of water. These properties, in turn, have different effects on the three hydrological services: (i) total surface and groundwater yields, (ii) seasonal distribution, and (iii) water quality (e.g. sedimentation). The exact relation between forest cover and these services is highly site-specific, and sometimes contested. Bruijnzeel (2004) concludes from a wide range of available scientific evidence that (a) total annual yield increases with the percentage of biomass removed, (b) infiltration is reduced by deforestation and subsequent soil degradation, thus often reducing dry-season flows; and (c) tree cover may prevent surface erosion and shallow landslides. Since total annual yield is of less concern in Costa Rica, we focus on (b) and (c) as prospective hydrological benefits from forest conservation. Our approach represents the interests of groundwater users, who hold 36% of water concessions and account for approximately 10% of total water volume used; and it represents the interests of those who seek low silt concentrations, including hydroelectric power stations, which account for 70% of total utilized water volume (Fallas, 2006).<sup>12</sup>

Ideally, hydrological service estimation would require information on site-specific soil characteristics, vegetation cover, slope, distribution and intensity of precipitation, as well as variable demand for different hydrological services. In our simplified estimation, we use slope as a proxy for erosion and sedimentation potential. Slope (%) was derived from an elevation model, drawing on contour lines from the Digital Atlas of Costa Rica 2004 (ITCR, 2004). Mean slope within the sample sites is 3.8% (Table 1). Sites with steep slopes thus receive higher forest conservation priority. Second, groundwater demand is determined using data by Imbach (2005) which aggregates water consumption (private, agricultural, and industrial) from all registered wells. Using an aquifer map from the same study, we can calculate per hectare water consumption for each aquifer. Water consumption from the sample sites has a mean of 16.2 l/h/ha (Table 1). Sites with high water use intensity will receive higher forest conservation priority. The z-values for slope and intensity of water use are combined to compute an aggregate mean z-value for hydrological services, giving equal weight to the two sub-criteria.<sup>10</sup>

### 3.4. Scenic beauty

We define scenic beauty services as the contribution of conserved forests to an aesthetically appreciated landscape. Deforestation often deteriorates the landscape vista, and thus its scenic beauty. Ideally, scenic beauty services would be measured as a function of the composition of various landscape elements. Thus, a specific plot's marginal contribution would depend on its spatial relation to other landscape elements. Moreover, service values would depend on the number of people who view the landscape, and their individual level of appreciation. In our simplified approach, we assume that (i) any loss of forest cover reduces scenic beauty services, and that (ii) a plot's degree of scenic contribution depends on its visibility. We calculated the visibility of forest from hypothetical lookout points spaced in equal distances along the national road network. The calculation was based on a triangulated irregular network (TIN) elevation file derived from ITCR (2004). Pixels within the 107 sample sites can be seen from a mean of 9.0 viewpoints (Table 1).

### 3.5. PES participation costs

As mentioned, participation costs consist of opportunity, transaction, and protection costs. We calculated opportunity costs focusing on pastures as the most likely alternative to natural forest. Natural forest itself, we assume, produces no commercial income. This is because logging and timber sales from natural forests are prohibited by law, unless a management plan has been certified by Costa Rican authorities — which in recent years has almost never occurred. Illegal logging and timber transport are risky, and very few rule violations seem to occur in the study area. Our data also show that commercial benefits other than from timber commercialization are close to zero. Though prohibited, gradual land-use change through the elimination of forest undergrowth and smaller trees towards pasture with scattered shading trees is locally somewhat more frequent. Thus, the opportunity cost of maintaining forest is equal to the foregone optional net income from pastures. This is likely to be a slight over-estimation of opportunity costs for several reasons. First, we are not considering the cost of land conversion. Second, we calculate opportunity costs from average pasture profitability, ignoring that forests are generally found on economically marginal areas with lower potential pasture productivity. Third, family labor is not deducted from opportunity costs, assuming there is no readily available income alternative. Fourth, even plots with negative net returns are still paid the sum of their transaction and protection cost. Factors (i) to (iv) thus increase participation cost estimates and weaken the potential of flexible payments to increase PES program efficiency. The mean per hectare net return of all properties was found to be US\$56.09 annually (Table 1).

On the Nicoya Peninsula, the great majority of PES applications for small and medium sized land plots (<100 ha) is processed by intermediaries (J.A. Jiménez Fajardo, pers. comm., 2007), who handle all transactions such as paperwork, consultancy, preparing technical studies, and supervision. For these services, the intermediaries charge a maximum of 18% of the current payment, i.e. US\$7.20/ha (FONAFIFO, 2005), which we use as an approximation for transaction costs.

<sup>11</sup> Carbon quantities vary depending on type and management of pastures, including the number of trees. However, to ease the analysis we work with a single average only.

<sup>12</sup> Groundwater volume was estimated using data from Fallas (2006).

Applications for large land plots (>100 ha) are normally given to private forest engineers (*regentes*) who may offer lower per hectare prices. For these land plots, we calculate a transaction cost of 12%, i.e. US\$4.80/ha.<sup>13</sup> Finally, protection costs mainly consist of establishing firebreaks, fencing off cattle and public PES signposting. We estimate protection costs for every individual plot, based on our survey data.<sup>14</sup> The mean protection cost is US\$3.56/ha/yr (Table 1).

### 3.6. Deforestation risks

We use an index for deforestation pressures estimated by Pfaff and Sanchez-Azofeifa (2004). They econometrically calculated an average annual deforestation rate ( $r$ ) for different Holdridge life zones within each district, based on forest clearing data over time (1963, 1979, 1986, 1997, and 2000).<sup>15</sup> We used  $r$  to calculate deforestation probabilities for a period of 5 years ( $r_5$ ), the usual duration of PES contracts in Costa Rica:<sup>16</sup>

$$r_5 = 1 - (1 - r)^5 \tag{5}$$

The mean of  $r_5$  of the 107 forest sites is 3.86% (Table 1). Combining the five-year deforestation risk ( $r_5$ ) of pixel  $i$  with its service score for biodiversity ( $b_i$ ), hydrology ( $w_i$ ), carbon ( $c_i$ ) and scenic beauty ( $s_i$ ), the pixel's additionality  $e_i$  can be estimated using Eq. (3), giving:

$$e_i = r_{5i}(b_i + w_i + c_i + l_i) \tag{6}$$

As a neutral point of departure, Eq. (6) gives equal weight to all four services. This could be changed. For instance, in the wake of coming water user conservation fees (Pagiola, 2008-this issue), increased weight might be given to water services in relevant watersheds. An alternative approach could be to adjust weights to the relative importance of different funding sources and the services they are most interested in. Fig. 1 shows additionality on Nicoya Peninsula. The conservation of lighter-shaded areas provides less additionality than the conservation of darker areas. The 107 sample sites deliver an overall environmental service score of 230,563 (mean 61.8/ha) and, if put under PES protection, would provide an additionality score of 9212 (4.0%) with a mean additionality of 2.5 score units per hectare (Table 1). Their total

<sup>13</sup> We maintain the absolute per hectare transaction costs (instead of percentages) also for the flexible payment approach giving "cheap" sites a comparable disadvantage in competing for program entry.

<sup>14</sup> Firebreak costs were taken directly from survey data. Fencing costs were calculated multiplying per hectare fencing costs on pasture with the factor 0.1818. Signposting costs were estimated at US\$5 every 50 hectares.

<sup>15</sup> Note that Pfaff and Sanchez-Azofeifa (2004) do not differentiate between PES and non-PES sites. In theory, this may lead to an underestimation of deforestation. We believe that this effect will be small because (i) forest cover data goes back to well before the implementation of PES in 1997 (thus PES may only have had a risk-decreasing effect on the forest-cover data from 2000); (ii) the ratio of PES forest (256,521 ha in 2000 (Ortiz, 2002)) to total forest cover (2,557,370 ha in 1996/97 (Calvo et al., 1999)) is relatively small and high risk areas were not systematically targeted.

<sup>16</sup> We tested the correlation of hazard rates by Pfaff and Sanchez-Azofeifa (2004) and our field data opportunity costs. Correlation coefficients are not significant (Pearson 0.90, Spearman 0.76).

water score is 26,432 (mean 7.1/ha). If the 107 sites were to be compensated with their individual cost of PES participation, total expenditure would amount to US\$264,263, implying an additionality efficiency of 34.9/US\$1,000. The sample contains 57 sites (1309 ha) with per hectare participation costs smaller than the fixed payment of US\$40/ha.

## 4. Targeting approaches

We compare eight different targeting approaches for choosing among the group of 107 forest sites. Given an equal budget, the results will show which of the approaches delivers most additionality. An overview of the eight targeting approaches and their main differences is given in the upper part of Table 2.

In the 'Baseline' approach, we follow the selection procedure currently employed by FONAFIFO. This means we select all forest sites that (i) have a cost of participation smaller than the fixed payment of US\$40/ha, and (ii) lie within priority areas.<sup>17</sup> For comparability, the total computed expenditure in the Baseline approach will also serve as the budget limit ( $C_{budget}$ ) in all other approaches.

The 'FlexAdd' approach is our main and most complete targeting approach. It employs flexible payments and utilizes information on service score and deforestation risk to compute additionality. Forest sites ( $j$ ) with the highest ratio of total additionality ( $e_j$ ) to total cost ( $C_j$ ) are selected and total additionality is maximized using:

$$\text{Max}_{\delta_j} \sum_{i=1}^n \delta_i(e_i) \tag{7}$$

where  $\delta_j \in \{0,1\}$  is an indicator variable which takes the value of one if site  $j$  is selected and zero otherwise. The maximization is subject to the budget constraint:<sup>18</sup>

$$\sum_{j=1}^n \delta_j(C_j) \leq C_{budget} \tag{8}$$

Given our assumption that landowners participate as long as they receive at least their participation cost, with flexible payments it would be optimal for the implementing agency to choose a level of payment just equal to (or marginally above) participation costs. Thus, flexible payment levels are set equal to participation costs here.

The 'FlexScore' approach is similar to the 'FlexAdd' approach in that it employs flexible payments equal to participation costs. But it only utilizes information on service score (ignoring deforestation threat) and selects sites with the highest ratio of total score ( $u_j$ ) to  $C_j$  subject to budget constraint (8).

$$\text{Max}_{\delta_j} \sum_{j=1}^n \delta_j(u_j) \tag{9}$$

<sup>17</sup> As we determine farm location with a point, it may occur that parts of it lie outside priority areas.

<sup>18</sup> In all approaches (except Baseline) we maximize by manually ranking plots from highest to lowest values of the selection criterion (e.g. additionality/cost in FlexAdd approach) and selecting plots until budget is depleted.

**Table 2 – Results of targeting approaches (percentages of values compared to Baseline selection in brackets)**

	Baseline	FlexAdd	FlexScore	FlexWater	Flex	FixAdd	FixScore	FixWater
Payment	Fixed	Flexible	Flexible	Flexible	Flexible	Fixed	Fixed	Fixed
Budget limit	No	Yes						
Selection	Priority	Additionality/	Score/cost	Water	Mean cost	Mean	Mean score	Mean water
Criteria	Area	cost		score/cost		additionality		score
Total cost (US\$)	30,028 (100.00)	30,014 (99.95)	29,997 (99.90)	30,016 (99.96)	30,000 (99.91)	30,016 (99.96)	30,012 (99.95)	30,024 (99.99)
No. of sites	20 (100)	56 (280)	62 (310)	44 (220)	68 (340)	37 (185)	36 (180)	25 (125)
Area (ha)	750.7 (100.0)	1350.2 (179.0)	1423.3 (189.6)	1178.7 (157.0)	1441.7 (192.0)	750.4 (100.0)	750.3 (99.0)	750.6 (100.0)
Mean site size (ha)	37.5 (100.0)	24.1 (64.3)	23.0 (61.3)	26.8 (71.5)	21.2 (56.5)	20.3 (54.1)	20.8 (55.5)	30.0 (80.0)
Total WaterScore	6,900 (100.0)	10,301 (149.3)	11,194 (162.2)	15,931 (230.9)	10,952 (158.7)	8,591 (124.5)	8,267 (119.8)	9,618 (139.4)
Total Env. Service	52,148 (100.0)	94,829 (181.8)	98,259 (188.4)	82,289 (157.8)	96,421 (184.9)	57,156 (109.6)	57,770 (110.8)	52,444 (100.6)
Score								
Total Additionality	1969 (100.0)	4033 (204.8)	3909 (198.5)	3211 (163.1)	3798 (192.9)	2294 (116.5)	2253 (114.4)	2088 (106.0)
Additionality/	65.6 (100.0)	134.3 (204.7)	130.3 (198.6)	107.0 (163.1)	126.6 (193.0)	76.4 (116.5)	75.1 (114.5)	69.5 (105.9)
1000\$								

Source: Own computations.

The ‘FlexWater’ approach is similar to the ‘FlexScore’ approach in that it employs flexible payments and utilizes service scores, but it only uses scores for water services. It selects the sites with the highest ratio of water score ( $u_{wj}$ ) to  $C_j$  (10) and budget constraint (8) applies. The ‘FlexWater’ approach simulates a funding situation for water services only. Other services are delivered as by-products.

$$\text{Max}_{\delta_j} \sum_{j=1}^n \delta_j (u_{wj}) \quad (10)$$

The ‘Flex’ approach employs flexible payments and simply targets the “cheapest” sites with the lowest per hectare participation costs (regardless of environmental services provided) until the budget is depleted (constraint 8 applies).

In the ‘FixAdd’ approach, a fixed payment of US\$40/ha ( $C_{\text{fix}}$ ) is used. It uses information on service score and deforestation risk to compute additionality. We select among the land plots with  $C_j/a_j < C_{\text{fix}}$  (where  $a_j$  is the size of plot  $j$  in hectares) those with the highest mean additionality values ( $\bar{e}_j$ ) and maximize total additionality (7) subject to the budget constraint:

$$\sum_{j=1}^n \delta_j (a_j C_{\text{fix}}) \leq C_{\text{budget}} \quad (11)$$

In the ‘FixScore’ approach we proceed as in the ‘FixAdd’ approach, but ignore deforestation threats. Payments are fixed, but we target the highest mean scores ( $\bar{u}_j$ ) subject to budget constraint (11) and maximize using (9).

The ‘FixWater’ approach is similar to the ‘FixScore’ approach. Payments are fixed but the sites with the highest mean water scores ( $\bar{u}_{wj}$ ) are selected. Like the ‘FlexWater’ approach, the ‘FixWater’ approach simulates a funding situation for water services only, with other services delivered as ‘by-products’.

## 5. Results

Table 2 presents the principal results from the eight targeting approaches. In the Baseline approach, 20 sites with a total area of 750.7 ha are selected. Total payments equal US\$30,028,

which also determines the budget for all other approaches. The Baseline approach yields a total overall score of 52,148 and a total additionality of 1969, which translates into an additionality efficiency of 65.6 per US\$1000. The total water score is 6900.

In each of the other approaches, total payments are almost identical, but always slightly below those of the Baseline because only entire sites can be selected, and hence small parts of the budget remain unspent. Similarly, the contracted area in the fixed payment approaches is almost identical. However, in the flexible payment approaches it rises sharply, by up to 192%. This is because average per hectare payments are lower, although individual payments go up to US\$61.95 in the FlexScore approach, US\$68.32 in the FlexAdd approach and US\$183.30 in the FlexWater approach.

All approaches achieve higher efficiencies (i.e., additionality per US\$1000 spent) than the Baseline approach, independent of whether additionality, overall score, or water score was targeted, but the degree to which efficiency increases differs. In general, increases in the fixed payment approaches (by 6 to 17%) are not as high as in the flexible payment approaches (increases of 63–105%) reflecting the dominance of cost-wise payment differentiation. The main reason for this is the considerable increase of contracted area in the flexible payment approaches. However, in some cases benefits increase more than area. For example, in the FlexAdd approach area increases by 79% while efficiency more than doubles, increasing by 105%. Therefore, the efficiency increase must have two causes: (i) targeting low-cost sites and thus increasing area, and (ii) targeting high additionality sites. To see how big the latter effect is, we compare results of the FlexAdd and Flex approach. We find that efficiency in the Flex approach, which simply maximizes total area by allowing for flexible payments, increased by 93%, compared to an increase of 105% in the FlexAdd approach. The remaining 12% may thus be attributed to targeting high additionality sites. Whether this relatively small incremental efficiency gain justifies the more complex targeting procedure of the FlexAdd approach is a subject that requires debate. In general, whether a policy of simply targeting the cheapest sites as in the Flex scenario leads to desired outcomes depends on the variability and correlation of

the desired services and the costs of providing them. For example, if only water services are of interest, the Flex approach increases water score efficiency by 59% compared to the Baseline approach, while targeting water services explicitly (in the FlexWater approach) yields an increase of 131%.

Even though efficiency is highest in the FlexAdd approach, it can also be seen that efficiencies in the FixAdd (76.4) and FlexAdd (134.3) approaches are only slightly superior to those in the FixScore (75.1) and FlexScore (130.3) approaches, by 1.7% and 3.1%, respectively. The reason for this is the low variability of deforestation rates ( $r_s$ ) within the sample sites, causing almost identical selection results between the score and additionality approaches. Additionality in the targeting approaches ranges from 3.8% of total service score in the Baseline to 4.3% in the FlexAdd. For example, the FixAdd selection has a total score of 57,156 and an additionality of only 2294, or 4.0%.

Compared to the Baseline approach, the FixWater and FlexWater approaches are particularly successful in increasing the water score, by 39% and 131%, respectively. While water service acts here as an “umbrella service” which provides other services as mere by-products<sup>19</sup>, the overall score and additionality for all four services also increase, though to a lesser degree than in their counterpart multiple service targeting approaches. This makes the water-based approaches interesting alternatives particularly when funding is obtained predominantly for one service. In 2005, for example, Costa Rica added an earmarked watershed conservation fee to the existing water tariff. Once fully implemented, this fee will generate an estimated US \$19 million annually, of which 25% would be channelled through the PSA program (Pagiola, 2008-this issue).

We also observe that the Baseline approach contracts sites with a mean size (37.5 ha) higher than any other approach (20.3–30.0 ha).<sup>20,21</sup> The selection tool’s bias towards smaller parcels thus made us ask whether this could imply increased participation of poorer households with less land available for program enrolment. Using monthly household consumption data from our survey as a proxy for wealth, we find that mean household consumption in the Baseline is higher (index 2.7), though not significantly, than in the other scenarios (index 2.4–2.5). This rough indicator thus may suggest that the targeting tool tends to select more lands belonging to poorer households. Higher opportunity costs, tenure insecurity, lack of land titles, technical constraints and transaction costs may be prime obstacles obstructing poorer landowners’ access to PES programs (Pagiola et al., 2005). Our results suggest that some poorer landowners can actually be more competitive service providers, providing higher environmental benefits per monetary unit spent. Policy

measures to reduce obstacles like the ones mentioned above are therefore justified not only on grounds of equity but also of program efficiency. However, note that in the case of flexible payments as presented here in the “Flex” approaches, payments only just compensate participation costs, implying that increased participation would not lead to increased welfare and poverty alleviation.

## 6. Sensitivity analysis

This analysis necessarily had to rely on relatively crude data and sometimes arbitrary assumptions. To what extent do the results remain stable if these assumptions were changed? To answer this question, we conduct a sensitivity analysis, in which we test how the selected sets of sites from the various targeting approaches perform if spatial service distribution differed due to alternative assumptions and scoring techniques. Imagine, for example, a situation where new data change the previous assumptions about service distribution. Our sensitivity analysis examines to what extent the originally selected sites’ service potential changes in terms of total score and additional score (additionality). We change our originally assumed multiple service distribution to four alternative distribution scenarios (i–iv), in which (i) a zero value for scenic beauty services is assumed for all sites, (ii) biodiversity scores depend exclusively on the representativity criterion and not on connectivity, (iii) the distribution of water services depends only on water use intensity and not on slope, and (iv) in which all three of these variations are combined. Moreover, for the single-service case, we examine how the parcel selections of the water approaches perform in terms of water score if assumptions about water distribution change: in addition to the original water service distribution, we now introduce distribution scenarios where only water use intensity or slope are considered.

We also examine the impact of using different scoring techniques. Instead of the z-normalization, we use an interval normalization (Ferraro, 2004) returning values from zero to one for each service and test how the originally selected sites score if multiple services were measured in interval-normalized scores. For large values preferred to low values, the interval score ( $k$ ) is computed as

$$k_i = \frac{x_i - \min(x)}{\max(x) - \min(x)} \quad (12)$$

where  $x_i$  is the observed value of a specific variable of pixel  $i$ ,  $\min$  is the minimum and  $\max$  the maximum value observed within the sample. For small values preferred to large values the normalization takes the form

$$k_i = \frac{x_i - \max(x)}{\min(x) - \max(x)} \quad (13)$$

The results of the sensitivity analysis are presented in Table 3. There is no direct comparability between the scenarios, because in each the entire scoring dimension changes. We therefore compare the results of the Baseline site selection in one service distribution scenario with the results of other targeting approaches in the same scenario.

<sup>19</sup> See Turpie and Blignaut (2008-this issue) for an example of how using water as an “umbrella service” can help achieve conservation goals in South Africa.

<sup>20</sup> The selection procedure of only allowing entire sites into the program results in somewhat smaller means, compared to a procedure allowing for the splitting of sites. However, we checked the splitting procedure and found that average site sizes (22.7, 23.5, 31.3, 24.5, 25.2, 26.8 ha) also stay well below the Baseline (37.5 ha).

<sup>21</sup> Note that we did not consider per-contract transaction costs to service buyers in the selection procedure. As these are likely to be independent of land size, their consideration in the selection procedure might discourage the inclusion of smaller sites.

For ease of comparison, the first column of Table 3 repeats the results from Table 2 (only those for interval normalization are new). In the results using z-normalized scores, a general pattern can be observed throughout the alternative service distribution scenarios: there is a low-to-moderate increase of total score from the Baseline to the FixScore approach; total score almost doubles in the FlexScore and stays slightly below FlexScore levels in the Flex approach. When additionality is derived by multiplying z-scores by deforestation risk, the same pattern can be observed for the Baseline, FixAdd, FlexAdd, and Flex approaches under all the alternative service distribution scenarios. When interval-normalized scores are used, the results also broadly follow this pattern, except that some of the total scores in the Flex selections are slightly higher than those in the FlexScore selection. We can conclude from these observations that any of the proposed targeting mechanisms increases bundled services also under changing assumptions and scoring techniques. However, for the fixed payment approaches the increases under some alternative assumptions may be too low to justify the use of a complex targeting system. Flexible payment approaches, on the other hand, are found to be clearly dominant even under alternative service distributions.

The water scores do not quite follow the above pattern. Drastic water score differences between the Baseline and FixWater approaches, as well as between the Flex and FlexWater approaches show the importance of well-defined targets in the presence of high service variability as is the case for water services. The bundling of service scores, on the

other hand, leads to spatially more evenly distributed services with less variability and therefore have more robust results.

## 7. Discussion

The flexible payment schemes were shown to yield the most significant increase in environmental service delivery and efficiency. In order to implement flexible payments in a PES program, precise estimates of participation costs are required. This study examined these costs in personal one-to-one interviews, but this is costly, and could cause landowners to answer strategically, thus producing biased estimates. One possible, less costly approach is to use easily observable spatial data as proxies for participation costs. The extent to which such an approach is cost-effective depends largely on how precisely it can estimate participation costs. Even if there was a cost-effective method to determine monetary costs, restricting the analysis to monetary values, as was done in this study, fails to include the non-monetary attitudes of the landowners, e.g. risk behaviour, mistrust, or conservation preferences. These attitudes could influence the level of the landowner's real willingness to accept (WTA).

Inverse auctions present an alternative that could take such subjective attitudes into account. In principle, inverse auctions aim to induce property owners to reveal their real WTA. However, in practice this is not always achieved (see Ferraro, 2008-this issue, for a detailed discussion). There are examples of the successful use of auction systems in

**Table 3 – Scoring results in the sensitivity analysis (percentages of values compared to Baseline selection in brackets)**

Targeting approaches	Alternative service distribution scenarios				
	Initial assumption	Omitting scenic beauty	Basing biodiversity only on representativity	Basing water only on groundwater	Combing all three changes
<i>z-normalized scores</i>					
Baseline	52,148 (100.0)	48,477 (100.0)	38,147 (100.0)	45,553 (100.0)	27,823 (100.0)
FixScore	57,770 (110.8)	49,414 (101.9)	43,602 (114.3)	51,734 (113.6)	29,151 (104.8)
FlexScore	98,259 (188.4)	86,490 (178.4)	74,460 (195.2)	90,416 (198.5)	54,736 (197.7)
Flex	96,421 (184.9)	84,508 (174.3)	72,600 (190.3)	88,851 (195.0)	53,003 (190.5)
<i>z-normalized additional scores (additionality)</i>					
Baseline	1969 (100.0)	1827 (100.0)	1440 (100.0)	1718 (100.0)	1044 (100.0)
FixAdd	2294 (116.5)	1928 (105.5)	1762 (122.4)	2074 (120.7)	1173 (112.4)
FlexAdd	4033 (204.8)	3511 (192.2)	3050 (211.8)	3748 (218.2)	2237 (214.3)
Flex	3798 (192.9)	3317 (181.6)	2873 (199.5)	3514 (204.5)	2104 (201.5)
<i>Interval normalized-scores</i>					
Baseline	15,182 (100.0)	14,412 (100.0)	13,197 (100.0)	14,859 (100.0)	12,104 (100.0)
FixScore	16,185 (106.6)	14,470 (100.4)	14,280 (108.2)	16,039 (107.9)	12,421 (102.6)
FlexScore	28,866 (190.1)	26,448 (183.5)	25,927 (196.5)	28,715 (193.2)	23,358 (193.0)
Flex	28,700 (189.0)	26,251 (182.1)	26,002 (197.0)	28,566 (192.2)	23,419 (193.5)
<i>Water-only scenarios</i>					
	Initial assumption	Basing water only on groundwater		Basing water only on slope	
<i>z-normalized scores for water services only</i>					
Baseline	6900 (100.0)	331 (100.0)		9648 (100.0)	
FixWater	9618 (139.4)	2312 (698.5)		12,223 (126.7)	
FlexWater	15,931 (230.9)	9190 (2776.4)		13,909 (144.2)	
Flex	10,952 (158.7)	3430 (1036.3)		13,020 (135.0)	

Source: Own computations.

developed countries such as the United States (e.g. Claassen et al., 2008-this issue), but no such examples exist for developing countries. Technically, it appears that an auction system could easily be integrated into most currently practiced PES programs. If landowners have to apply formally for program participation (as is the case in Costa Rica's PSA Program), their bid could be part of the application process. Auction systems might also be a powerful way of making payment differentiation politically acceptable, because service sellers suggest the price themselves. The applicability, impediments, and challenges of auction systems for PES programs in developing countries certainly deserve further attention in future research (Ferraro, 2008-this issue).

Our proposed targeting mechanism would increase the service buyer's transaction costs — a factor not considered in the above. In Costa Rica, FONAFIFO's transaction costs are limited by law, implying that increasing transaction costs would require savings elsewhere. There are two main sources of incremental transaction costs associated with our suggested targeting approach: (i) changes in administrative processes; and (ii) the cost of creating and maintaining the targeting tool itself. For (i), we identify two significant changes: First, site coordinates of all applicant sites have to be fed into the selection tool. As digitization is already undertaken for all approved sites (755 in 2005, FONAFIFO, 2006), this only imposes additional costs for rejected sites — about 1510, assuming there are about three times more applications than signed contracts (E. Ortiz, pers. comm., 2004). At a labor cost of US\$50/day, we estimate the additional costs to be about US\$2,400/year. Second, the new targeting approach would change the seasonal workload distribution, because applications can only be fully processed after the site selection is completed. Theoretically, this should not impose additional costs, and is thus not specifically considered here.<sup>22</sup> Regarding (ii), we estimate that tool creation, maintenance, data update, and continued data improvement could be realized by a full-time GIS expert with an annual gross salary of about US\$30,000 and overhead and equipment costs of around US\$10,000.<sup>23</sup>

Total additional transaction costs should therefore be less than US\$42,500/yr. For 2006, the program's total budget was US\$15.22 million, including a politically determined administrative budget of US\$1.12 million (MINAE, 2006). Thus the estimated cost for the proposed targeting mechanism would represent 0.28% of the total budget and 3.8% of the administrative budget. If we assume that actual environmental service payments budgeted for 2006 (US\$ 14.1 million) reflect a minimum valuation of the services delivered, an efficiency increase of e.g. 14.4% in the FixScore approach would correspond to an increase of services of US\$2.0 m (relative to Baseline efficiency). The cost-effectiveness of using the new tool therefore seems to stand beyond any doubt.

<sup>22</sup> FONAFIFO used to process all applications at once, but switched to a rolling process which more evenly distributes workloads. We acknowledge that FONAFIFO might not agree with our view on cost neutrality.

<sup>23</sup> The initial costs of tool creation might be much higher, but this cost could probably be supported by a donor, and so is possibly less of a constraint than the increase in annual operating costs.

Since 2006, the PSA program also includes payments for the retirement of agriculturally used land for natural forest re-growth. The tool could also help target regeneration contracts. To do so, it would need probability estimates on the likelihood of land retirement in the absence of PES. This retirement process can be observed in areas with decreasing productivity, increasing input prices, or decreasing product prices. In the presence of land abandonment, the additionality of a PES program then does not only depend on avoided deforestation through PES, but also on induced land retirement through PES.

## 8. Conclusions

Using data from the Nicoya Peninsula, we showed that the financial efficiency of PES programs like Costa Rica's PSA program could potentially be substantially increased by using a targeting process that integrates spatial data rather than a targeting system based solely on priority areas. All the targeting approaches considered led to higher efficiency outcomes, compared to the Baseline approach. However, the spatial attributes (benefits, risks, and costs) contribute very differently to the efficiency increase: While the integration of environmental service scores led to moderate efficiency increases, the integration of participation costs boosted efficiency significantly, largely due to lower average payments and, consequently, greater total contracted area. The use of deforestation probabilities, on the other hand, barely improved efficiency in this case, as this attribute shows little variation between sites in the study region. Nevertheless, in each country or region the potential efficiency impact of each spatial attribute depends largely on its variation in space and its correlation to other attributes, which may differ across regions and also with scale. Deforestation rates, for example, might be the key criterion in countries or regions with higher and regionally more variable deforestation threats than observed in Costa Rica.

Watershed protection is often the only fund-generating environmental service and may therefore, in practice, often function as an 'umbrella service'. Focusing selection on water services only, gave water services a sharp increase and the overall environmental service efficiency considerably increased in the flexible payment approaches — though to a far lesser extent than in the multiple service approaches. Hence, if multiple service provision is the goal, the water-only selection cannot compete with the integrated service-selection approaches. However, if water users are the only service buyers, a single-service focus is a legitimate option that still offers some side benefits.

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