International Payments for Biodiversity Services: Review and Evaluation of Conservation Targeting Approaches[#]

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Abstract

The lack of incentive flows between local producers of global biodiversity-related ecosystem services and global service beneficiaries calls for a mechanism to avoid the under-provision of biodiversity services. Payments for ecosystem services (PES) are increasingly being implemented on the local and national scale. The issue of how PES can be upscaled to an international level is currently being discussed. International PES (IPES) will most likely be confronted with a limited budget and attention will have to be given to the issue of how payments are most effectively allocated. The objectives of this paper are to (i) outline the principles of targeting in conservation, (ii) provide an overview of techniques applied in science and practice, (iii) identify some of the specific challenges of an IPES scheme and (iv) analyze the suitability of available global targeting mechanisms for utilization in IPES. The paper is based on a review of targeting literature and uses the framework of a multi-criteria analysis to help organize and quantify strengths and weaknesses of alternative global targeting approaches. Despite growing consensus on the importance of incorporating costs, none of the global targeting approaches under review have so far incorporated costs as a targeting criterion. Data availability is probably one of the main constraints of global targeting. A stepwise selection approach could partly overcome this problem. Existing global targeting approaches could be used for first step selection choices. We identified Biodiversity Hotspots, Crisis Ecoregions and Endemic Bird Areas to be the most suitable approaches for IPES.

Key words: Payments for Ecosystem Services, Payments for Environmental Services, Targeting, Biodiversity, IPES, Conservation, Multi-Criteria-Analysis

1. Introduction

Ecosystems provide services on different spatial scales (MEA 2005). For example, tropical forest ecosystems conserve in-vivo genetic material, an ecosystem service which benefits the global community. Hydrological services, on the other hand, occur on local to regional scales depending on the size of the watershed in question. In the absence of effective incentive flows between scales “local efforts to provide ecosystem services are unlikely to consider global benefits, and global beneficiaries are prone to free-ride on local efforts” (Farley et al. 2010, p. 2075). Consequently, there is a need for a global institution in order to overcome the likely under-provision of global services. Payments for ecosystem services (PES) may be an effective instrument to complement international agreements where these do not offer sufficiently far-reaching or timely responses. PES can be far more cost-effective than indirect incentives (Ferraro and Simpson 2002) and interest in PES has grown considerably in recent years (Pattanayak et al. 2010). Carbon services have been subject to the development of international incentive markets (e.g. Clean Development Mechanism, CDM, or the currently debated mechanism to Reduce Emissions from Deforestation and forest Degradation, REDD). For biodiversity services, however, no such international mechanism exists - despite the global challenge of ecosystem decline (Chilchinisky & Proctor 2008). So far, we only find isolated PES initiatives. Numerous NGOs, for example, channel payments across countries to provide global existence values in selected project sites. The Global Environmental Facility (GEF) has also made direct payments for global biodiversity-related ecosystem service provision (e.g. Ecomarkets project in Costa Rica). Yet, the majority of GEF projects provide indirect payments which are, though ex-post evaluated, non-conditional on outcome. For these reasons the necessity for an up-scaled and biodiversity focused international PES mechanism (IPES) has been articulated (Hubermann 2009; Chilchinisky & Proctor 2008; Farley et al. 2010).

IPES could take many forms. A cap-and-trade mechanism, for example, would obligate countries to conserve biodiversity and facilitates the trading of biodiversity credits. But it faces the difficult-to-resolve challenge of standardizing a highly complex, diverse and location-specific product[1]. We therefore focus our discussion on an international IPES fund as a more conventional and possibly more acceptable instrument. A fund could be connected to other payment mechanisms such as REDD+ directing carbon mitigation efforts to forests (or non-forest landscapes in the case of REDD++) with exceptionally high biodiversity services by making additional financial incentives available (Harvey et

[1] Local credit based offset initiatives do exist as compiled in Madsen et al. (2011)

al. 2009, Karousakis et al. 2009, Busch 2010). The financing of such a fund could have several potential sources. One of the central issues, independent of the source of funding, is the question of how funds should be distributed to the many possible sites around the planet. Limited budgets force conservation agents to critically examine where and how scarce funds should be allocated, also referred to as targeting (Babcock et al. 1997) or prioritization (Brooks et al. 2006).

The objective of this paper is to provide an overview of existing targeting approaches and evaluate their strengths and weaknesses for their application in IPES. We focus on the targeting of ecosystem services which are primarily related to the biodiversity components of an ecosystem and we term these ‘biodiversity services’. Often the sensitivity of biodiversity services to variations in biodiversity is often poorly understood (Barker et al. 2009). For example, although soil organisms are acknowledged to play an important role in water purification, it is not entirely clear to what extent changes in their diversity affect this role. Yet, for other services the dependence on diversity is clear and strong. Ecosystems provide a service by sustaining in-vivo genetic diversity which inevitably declines with the biodiversity of an ecosystem. Genetic diversity is the central resource for benefits from bioprospecting, it holds existence values and it contributes to benefits in industries that are based on ornamental species. An ecosystem’s maintenance of genetic diversity is therefore a ‘final’ ecosystem service which Boyd and Banzhaf (2007, p.619) define to be a ‘component[s] of nature, directly enjoyed, consumed, or used to yield human well-being’. Following an argument by Boyd and Banzhaf (2007) we often refer to biodiversity per se (biodiversity stock) as a proxy for biodiversity services acknowledging, however, that the two are not always perfectly correlated. For illustrative reasons, we also use examples of services which are not primarily related to biodiversity. Biodiversity focused conservation can also provide considerable benefits in terms of other ecosystem services such as carbon storage and freshwater services (Larsen et al. 2011), but the discussion of such synergies is beyond the scope of this paper.

The paper is structured as follows: in section 2 we provide an overview of conservation targeting principles as well as brief reviews of available targeting approaches and techniques. The section is divided into four subsections following the three main targeting criteria 'service levels' (2.1), 'costs of service provision' (2.2) and 'non-provision probability' (2.3), as well as the issue of how criteria can be integrated (2.4). This is followed, in section 3, by a discussion of specific challenges international payments for biodiversity services are likely to be confronted with. Section 4 provides an overview of the targeting characteristics used in currently available global targeting approaches. Applying a multi-criteria-analysis, we score and rank the global targeting approaches according to their suitability for IPES in section 5. In the last section we conclude with some final remarks.

2. Principles of Targeting Payments for Biodiversity Services

Targeting is defined here as the process of directing payments towards locations where desired biodiversity services are produced most efficiently. Efficiency refers to the best possible service to cost ratio. A program manager's ultimate objective in targeting is to obtain a maximum of biodiversity services with a limited budget, or a defined amount of services at least cost.

Targeting techniques are far from unambiguous and therefore a wide range of targeting approaches on the local, national and international scale have been developed (e.g. Alix-Garcia et al. 2008; Barton et al. 2003; Drechsler 2011; Ferraro 2004; Hajkowicz et al. 2007; Hoekstra et al. 2005; McCarthy et al. 2008; Murdoch et al. 2007; Myers et al. 2000; Olson & Dinerstein 2002; Rodrigues et al. 2003; Rolfe and Windle 2011; Wünscher et al. 2008).

Targeting in PES programs would be less critical if it was possible to directly attribute the provision of services to individual land managers. If this was the case, payments could be made ex-post upon the delivery of a service (e.g. Zabel & Holm-Müller 2008; Zabel and Roe 2009; Zabel et al. 2011). But due to the complexity of ecosystem processes and the influence of factors beyond the landholder’s control, the exact service amount and the land manager’s role in providing these often cannot be quantified. Therefore, payments for ecosystem services are normally made on the condition that specified actions which are thought to contribute to the delivery of the desired service are implemented (Engel et al. 2008; Wunder et al. 2008)[2]. In this ‘action-based’ approach, however, the uncertainty of whether services will be provided is entirely on the service buyer’s side. It is therefore in her interest to make predictions about the magnitude with which a land use action on a specific site is likely to contribute to service provisions.

[2] See also Gibbons et al. (2011) who theoretically model the conditions under which each approach (action vs. performance) is more cost-effective.

Normally, these contributions differ spatially across a landscape and the service buyer would target payments accordingly.

In addition to estimates concerning biodiversity services two other targeting criteria need to be considered in order to make selection choices, namely provision costs and the probability that a service is not provided in the absence of payments – here simply referred to as ‘non-provision probability’. Services may not be provided because intact ecosystems are degraded or because degraded ecosystems are not improved in the absence of payments. The targeting literature often refers to the probability of the former as threat, vulnerability, pressure or risk. Wünscher et al. (2008) used the term ‘risk’ to refer to both situations . Here, we use ‘non-provision probability’ to avoid confusion with other meanings of risk . ‘Non-provision probability’ is a critical component for the calculation of the additionality of payments, a clear demonstration of which will crucially determine the future success of PES schemes (Wunder 2005). Additionality can be calculated by multiplying the ‘non-provision probability’ (a value between 0 and 1) with the service potential of a given location. In other words, making payments in regions with a ‘non-provision probability’ equal to zero means that additionality is also zero. On the other hand, payments in regions that will surely not provide desired services in the absence of incentives (non-provision probability equal to one) have an additionality equal to the service potential. Most targeting approaches in conservation consider the first criterion (service levels). A growing number of approaches include costs (e.g. Alix-Garcia et al. 2008, Balmford et al. 2000, Drechsler 2011, Wendland et al. 2010, Wünscher et al. 2008), and with the use of proxies ‘non-provision probabilities’ are often indirectly addressed, for example by considering the presence of endangered species in a region or the vicinity and density of human populations.

2.1 Targeting Areas with High Service Levels

Ideally, a monetary value should be put on service levels to allow a direct comparison of services between sites. However, low-cost valuation techniques do not unequivocally determine the monetary value of expected services (e.g. Balmford et al. 2011) and would require a reliable estimate of service quantities, which is generally difficult to provide. Therefore, service estimates are usually based on indicators or proxies (e.g. slope of a hillside) that significantly influence the extent to which a promoted action (e.g. increase of vegetative cover) will lead to a desired service (e.g. erosion reduction). Assuming that costs and 'non-provision probabilities' are homogenously distributed across a landscape, a PES program for the reduction of erosion, for example, is likely to enhance program efficiency by targeting parcels with steep slopes. This does not mean that vegetation removal on parcels with flatter slopes does not lead to erosion, but rather that the particular objective, ceteris paribus, can be achieved at a lower cost by targeting the steepest parcels. In general, the estimation of expected services requires several steps. First, the desired service needs to be defined precisely (e.g. reduce soil erosion). Second, appropriate land use actions that are known to support the provision of the desired service have to be found (e.g. zero tillage farming). Third, indicators that are known to be correlated with the effectiveness of the land use action to provide the service need to be identified (e.g. slope).

2.2 Targeting Areas with Low Costs

The objective of including costs in targeting is to allocate payments where costs are lowest relative to service levels and ‘non-provision probabilities’. Program expenditure can be significantly reduced by adjusting payments to correspond with the actually incurred costs of service provision. Unlike biodiversity information, global data sets on conservation costs are scarce (Balmford et al. 2000). A notable exception is a global review by James et al. (1999) who reviewed management costs of protected areas. Balmford et al. (2000) complemented this information with country level land values and cost estimates for closing conservation gaps. Management costs of protected areas were also the focus of Moore et al. (2004) who applied this approach to Africa. This information was used by Wilson et al. (2006) to derive a regression model for African conservation management costs. Wilson et al.’s data on management costs of protected areas may not be suitable as an estimate for opportunity costs in an IPES scheme because IPES should include privately owned or managed land as conservation targets. But the methodology could be adapted and applied to estimate opportunity costs. A more recent dataset was developed by Naidoo and Iwamura (2007) who produced a global map of gross economic rents from agricultural lands and therefore provide data directly applicable to IPES. Their estimates are derived from data on crop and livestock distribution, yields and producer prices. Similar data would also be required for forest and aquatic systems. Naidoo and Iwamura’s approach has some shortcomings: (i.) they estimate gross rents, while net values would be more relevant, and (ii.) their estimates represent the costs of complete land retirement, while some desired land use actions would allow for partial utilization of the land with somewhat smaller opportunity costs.

Recent developments in the REDD negotiation process have stimulated new studies to estimate the cost of forest conservation. Grieg-Gran (2009), for example, provides national scale estimates for the cost of avoiding deforestation in eight countries accounting for 46% of global annual deforestation; and Rametsteiner et al. (2009) and Sohngen (2009) computed the cost of avoided deforestation in global models.

When it comes to choosing between specific land parcels on the local scale, global or regional estimation approaches are too unspecific. Estimating site-specific costs on the micro scale, particularly opportunity costs, can be challenging as there may be a large variation in profitability across sites. Furthermore, landowners may act strategically in reporting costs, and a number of factors which are difficult to measure, such as personal risk considerations or preferences, may influence individual opportunity costs (Wünscher 2009). The main approaches for estimating micro-level opportunity costs in practice include using land values (Chomitz et al. 2005), computing farm budgets (Wünscher et al. 2008), inferring values on the basis of economic and environmental data (similar to Wilson et al. 2006), and applying auctions to identify land owners' minimum willingness to accept (WTA) for including a site in a program (Ferraro 2008). In addition to opportunity costs the WTA includes transaction and protection costs, if applicable, as well as personal preferences, and is therefore a more relevant measure to determine payment levels.

Cost-aligned or flexible payments for efficient utilization of program budgets may face political barriers. Fixed payments, that is all landholders receive identical payments per unit of area, are often perceived to be fairer and more transparent by landowners and policy makers alike because they are uniform and pre-determined (Ferraro 2008). Although the voluntary character of PES makes it unlikely that landowners participate in PES programs to their disadvantage, it is often argued that flexible payments do not take equity into account sufficiently. This is probably the reason why most existing larger PES programs in developing countries employ either strictly fixed payment approaches such as the PSA program in Costa Rica (Pagiola 2008), or only allow very restricted payment variations with two or three different payment levels, such as the PSAH program in Mexico (Muñoz-Piña et al. 2008) and the “Grain for Green” program in China (Bennett 2008). However, although fixed payments are nominally equitable in that they pay every landholder the same rate, they can yield high rents (i.e. the difference between payment and incurred cost) to low-cost environmental service providers and low rents to high cost providers. As it is not the nominal rate but the rent that matters to the landholder, differentiating payments by provision costs could in fact produce more equitable outcomes than fixed payments.

2.3 Targeting Areas with High Probabilities that Services are not Provided in the Absence of Payments

If multiplied with the potential service level of an area the ‘non-provision probability’ helps to measure the additionality of payments, i.e. the service that is delivered beyond the respective baseline or business-as-usual scenario. A clear demonstration of additionality will crucially determine the future success of PES schemes (Wunder 2005). As Hartshorn et al. (2005, p.12) put it in the context of forest conservation: “Paying for forest protection on land that requires no protective measures is an inefficient use of scarce conservation funds". The additionality of prominent PES programs such as that in Costa Rica has been highly debated (Arriagada et al. 2009; Hartshorn et al. 2005; Ortiz et al. 2003; Sanchez-Azofeifa et al. 2007) and several studies found it to be low (e.g. Sanchez-Azofeifa et al. 2007, Sierra and Russman 2006). There are good reasons, therefore, that 'non-provision probability' should also be a key targeting criterion in an IPES scheme. From a global perspective, some of the highest 'non-provision probabilities' are likely to be found in developing countries, for example where deforestation is highest.

The probability of deforestation is often modeled using econometric analysis. In forestry, econometric models are by far the most widely used techniques to determine deforestation probabilities (Kaimowitz and Angelsen 1998). Examples are Pfaff and Sanchez-Azofeifa (2004), who econometrically estimated deforestation rates in Costa Rica, as well as Alix-Garcia et al. (2008), who regressed the deforestation rate for communal lands (ejidos) in Mexico (based on land cover data from 1994 and 2001) on a number of exogenous variables.

Instead of directly estimating 'non-provision probabilities', conservation targeting also makes use of proxies. Sanderson et al. (2002), for example, use population density, land transformation, accessibility and electrical power as proxies for the probability of habitat destruction. Some of these variables are likely to be part of the mentioned regression models, too, and may have been chosen because models frequently found these to be significant explanatory variables. Using proxies, however, does not provide quantifiable probabilities. IPES would clearly require quantifiable estimates for 'non-provision probabilities' for land use in general, including forestry and other ecosystems. Given today's availability of satellite data computing such estimates is unlikely to pose major obstacles to IPES targeting.

2.4 Integration of Criteria

If multiple indicators are chosen to describe a service and/or if multiple services are targeted, the issue arises of how they can be combined to adequately consider interactions and trade-offs (Drechsler 2011). Literature offers various integrating approaches to deal with the trade-offs of multiple indicators and objectives. A simple way of addressing this problem is the use of a stepwise approach as realized by Myers et al. (2000) for the identification of biodiversity hotspots: "A second determinant of hotspot status, applied only after an area has met the 'plants' criterion, is the degree of threat through habitat loss" (Myers et al. 2000, p. 855). Stepwise selection only requires a ranking of the attributes and objectives according to importance. Other approaches include the integration of different services with a weighted sum of standardized indices as in the Integrated Silvopastoral Ecosystem Management Project (Pagiola et al. 2007) or the US Conservation Reserve Program (Claassen et al. 2008); the non-parametric distance function approach (Ferraro 2004); normalization procedures to make indicators and objectives directly comparable (e.g. Wünscher et al. 2008); or program specific approaches such as the Australian Bush Tender's multiplication of a Biodiversity Significance Score, which depends on the type of plants and animals in a site, with a Habitat Service Score, which depends on the type of management commitment (Department of Sustainability and Environment 2008).

3. Challenges of IPES targeting

Due to data limitations on the global scale, a targeting scheme for international PES will need to use simple and easily accessible data that reflects the major interests of service buyers and is handled in the most transparent way. Although data availability has improved continuously in the last years (Rodrigues et al. 2003), biodiversity data on the global level is still incomplete and sometimes coarse. The availability of global information on ‘non-provision probabilities’, and especially costs is even scarcer. One way of dealing with data limitations on a global scale is to split the prioritization process into two or more steps. Such a stepwise approach would first identify global priority regions (first step) and then apply fine-tuned techniques within the regions requiring local targeting strategies (second and additional steps). This way, the targeting procedure can zoom in to land parcel units at the local or project level.

Though numerous studies have shown the merits of systematic targeting (e.g. Alix-Garcia et al. 2008; Ferraro 2003), elaborate approaches are rarely implemented in practice. Hajkowicz et al. (2007), for example, note that in the case of biodiversity planning no complete set of areas produced by computer algorithms has been implemented anywhere in real-world projects, in spite of a decade or more of work on reserve selection methods. It is possible that improved targeting mechanisms are not used in practice due to their complexity and their lower levels of transparency for policy makers and project proponents. Hajkowicz et al. (2007) therefore argue that the use of suboptimal project selection procedures might be justified, considering that they provide more transparency, accountability and auditability.

It is therefore unlikely that the ‘latest targeting technology’ will find consensus among stakeholders at an international level. In general, there is a trade-off between targeting optimality on the one hand, and simplicity, transaction costs, transparency and consensus on the other hand. Efficiency gains of improved targeting need to be compared to their implementation costs. If ‘optimal’ solutions have so far found little acceptance at the project level, their issues are likely to be stumbling blocks at the global level. While PES projects at the local level have the advantage that the interests of the individual service buyer can be incorporated into the targeting structure, thus reflecting the purchaser’s demands, the global level will have to create a compromise that sufficiently satisfies all parties.

Two targeting criteria that have not been discussed above but can affect conservation outcome at the global scale, are ‘readiness’ and ‘leakage’ (Murray 2009). Readiness is the overall capability to implement service provision measures with the standards set by the payment scheme. It includes, among others, political will, public awareness, stable institutions, technical capacities, good governance and legislation. From a global perspective, it might be more cost-effective to target payments to countries that already provide a certain level of institutional capacity. Yet, it is often the countries with weaker institutions that are associated with particularly favorable service level to cost ratios and high ‘non-provision probabilities’, e.g. due to high deforestation rates. The participation of such countries in IPES can also have a political dimension in that global equity is promoted. Dedicating a portion of an IPES fund to creating ‘readiness’, as currently pursued in the Forest Carbon Partnership Facility and the UN-REDD Programme, would be a potential way of addressing these issues.

Leakage refers to the shifting of service losses from areas where payments are allocated to uncontrolled locations. This negative leakage can undermine the effectiveness of payments. Leakage can also be positive but appears to be less frequent and substantial and may therefore not require as much attention in targeting as the adverse effects of negative leakage (Aukland et al. 2003). Leakage of tropical forest conservation, for example, depends on the factors underlying deforestation in the first place and the extent to which those factors are mobile (Murray 2009). Reducing agricultural expansion as a driving force for deforestation in one country might shift expansion to another country, particularly if land-clearing was for the purposes of cash crop production for global commodity markets. If, however, subsistence agriculture or local market production is the driver of a country’s deforestation, reducing deforestation there may not cause much international leakage, unless it heightens reliance on cash crops produced abroad (Murray 2009). To ensure more additionality of IPES, the concept of leakage could be incorporated into a global targeting mechanism, placing payments where economic models predict leakage to be less prevalent, or discounting the amount of ES provided by the estimated leakage.

The design of a global IPES targeting scheme will depend very much on the type of PES mechanism that will be applied. Computer based algorithms used for the selection of optimal sets of conservation projects, for example, produce adequate results more reliably if a uniform conservation measure and objective is pursued across projects. On a global scale, however, we encounter areas with very different requirements and site specific objectives and measures. One way of overcoming this is the mentioned stepwise targeting approach in which site specificity increases with every step.

4. Global Targeting Approaches

There already exist a number of well developed global targeting approaches which have been used to define global sets of priority areas for biodiversity conservation (Brooks et al. 2006). Brooks et al. (2006) describe nine such approaches, provide respective global maps and find that there is considerable overlap among some approaches, but also strong trade-offs between others. In Table 1 we describe the approaches by their targeting criteria and the techniques applied to integrate the criteria and indicators. The approaches provide a basis for first step priority setting in an international PES scheme.

[Table 1 about here]

Second step approaches are not discussed in more detail in this paper but it should be noted that a number of models exist to support second step selection choices such as Aries (Artificial Intelligence for Ecosystem Services, Villa et al. 2009) and Mimes (Multiscale Integrated Earth Systems model, Boumans and Costanza 2007) which address the flow of services between locations in addition to location specific provision of services as well as InVest (Integrated Valuation of Ecosystem Services and Tradeoffs, Nelson et al. 2009) which explicitly takes economic trade-offs of alternative land use options into account.

Some of the names of the global targeting approaches suggest differences in the pursued services (e.g. Centers of Plant Diversity or Endemic Bird Areas). But birds or plants serve principally as indicators, not as the service focus itself which rather is the entire range of biodiversity services associated with the conservation of the identified priority areas. Other approaches seem to contradict each other but are in fact complementary in that they target different aspects of biodiversity. The Last of the Wild approach (Sanderson et al. 2002), for example, argues for the conservation of vast wild areas that have remained largely intact as a pro-active approach to future threats. The Biodiversity Hotspot approach (Myers et al. 2000), on the other hand, is a rather reactive approach focusing on the most immediately endangered ecosystems that are rich in biodiversity. A holistic conservation approach would probably need to address both priority types. Nevertheless, IPES may be more suited for Biodiversity Hotspots, while vast areas with distant future threats are likely to have public as opposed to private land title and therefore could possibly be protected more efficiently using conservation instruments such as national parks. Table 1 is meant to give an overview of some of the global targeting approaches that are available to date. As none of the presented approaches have incorporated costs directly, a cost-effective IPES mechanism would need to complement any of the presented approaches with cost data.

5. Multi-Criteria Analysis of Global Targeting Approaches

In this section we undertake to identify the most suitable global targeting approaches (Table 1) for ‘first-step’ priority setting in an international PES scheme. Given the diversity of approaches and evaluation criteria we use the framework of a multi-criteria analysis to help organize strengths and weaknesses. The analysis is based on six criteria, namely transparency, low data requirements, focus on private land, use of indicators for 'non-provision probabilities', use of indicators for service levels, and flexibility. Use of costs is not part of the analysis as none of the approaches integrated cost estimates. Consideration would therefore not add variability between them. For each criterion we assign a score which takes the value of 'zero' if the criterion was not fulfilled, 'one' if the criterion was partly fulfilled, and 'two' if the criterion was indeed fulfilled. The approaches with the highest overall score are those which we find to be most suitable for IPES.

'Transparency' refers to the degree to which the approach is clear about the type of data it uses and the way the data are combined. Scores increase with transparency. 'Low data requirements' addresses the amount, diversity and complexity of data that is required. It also takes the availability of data into account. Low data requirements are preferable and receive higher scores. For the criterion 'focus on private land' we try to assess the extent to which the selected areas of an approach are likely to include private land. Scores increase with likeliness. The motivation for this is that payments for ecosystem services have a focus on private land tenure. The 'Use of indicators for non-provision probabilities' refers to if and how the approach incorporates estimates for 'non-provision probabilities' and the score increases with the suitability of the estimate for the calculation of additionality. The 'Use of indicators for service levels' refers to if and how service estimates are integrated into the approach. The last criterion considers the 'flexibility' with which the approach can be adapted to integrate additional or other variables such as cost data.

We acknowledge the limitations of a multi-criteria-analysis in general, and our choice of criteria and scoring decisions in particular. We are aware that scores could be allocated differently to possibly other criteria based on alternative lines of arguments. Nevertheless, we believe that the results can help policy makers, conservation practitioners and scientists develop a clearer view of the advantages and disadvantages of the presented approaches.

The results of the multi-criteria-analysis are summarized in Table 2. Most global prioritization approaches are transparent in the type of utilized data and the way they are combined. None of the approaches use complex computerized optimization procedures. Therefore, all but three approaches receive a score of two. Centers of Plant Diversity and Frontier Forests both stand out with a score of zero. Due to their reliance on expert knowledge decision making in the selection process is not reproducible (Brooks et al. 2006). Even if the decision making process was well documented (the respective reports on Forest Frontiers and Centers of Plant Diversity do not provide such documentation) the complexity of argumentation is likely to be less comprehensible for service buyers so that overall transparency is reduced. It is important to point out that the transparency criterion does not judge how well the approaches succeed in identifying areas that suit their declared objectives. Expert workshops can generate reliable results but they reduce transparency to outsiders. Megadiversity Countries utilizes concrete measures for endemism and a knock-out criterion for countries without marine ecosystems[3]. But other criteria are included less rigorously, e.g. ecosystem diversity. The approach therefore only receives one point.

[3] A country without a marine ecosystem does not qualify as a Megadiversity Country, independent of the diversity in other ecosystems.

[Table 2 about here]

‘Data requirements’ are relatively low for four approaches. Crisis Ecoregions, for example, depend on three spatial variables only (ecoregions by Olson et al. 2001, land use, protected areas), the data of which appear to be relatively accessible through remote sensing and national or international protected area registries. The advantage of Endemic Bird Areas is that information on bird distribution and threat levels is relatively well documented so that the complexity of utilized data is limited. Together with Wilderness Areas and Last of the Wild these approaches were given full points. On the other hand, Centers of Plant Diversity and Frontier Forests, which were assigned zero points only, not only have relatively large data requirements for the diversity of variables used but they also depend on data generation in expert workshops which, globally, appears to be a major undertaking for every update that may become necessary. With respect to ‘low data requirements’ the disadvantage of Global 200 Ecoregions (one point) is rather to be found in the type of required data such as unique ecological and evolutionary phenomena which are less clearly defined and accessible.

With scores of zero, Wilderness Areas and Last of the Wild stand out considering the criterion ‘focus on private land’. Seeking untouched and unthreatened areas, few of the selected regions are likely to be subject to private tenure and PES may not be the right instrument for these areas. The Last of the Wild approach, for example, includes the Saharan Desert as a priority area, a region in which property rights are not well established and monitoring would be costly also in practical terms. Endemic Bird Areas, Megadiversity Countries and Global 200 Ecoregions receive scores of one because privately owned land is expected to be selected randomly, rather than due to the defined targeting criteria. The focus on land conversion in the remaining targeting approaches is likely to shift selection choices to privately owned land (albeit not only). Crisis Ecoregions, Biodiversity Hotspots, Centers of Plant Diversity and Frontier Forests therefore all receive a score of two.

Seven approaches use measures for ‘non-provision probabilities’ in their set of targeting variables. However, two of these (Wilderness Areas and Last of the Wild) use the measures to identify areas in which the probability that services are not provided in the absence of payments is low which is the opposite of what we postulated ‘non-provision probability’ to show. The two approaches are therefore given zero points. Megadiversity Countries and Global 200 Ecoregions were also given a zero score for not using any measure for ‘non-provision probabilities’ at all. The remaining five approaches receive full points.

All but two approaches use service proxies in formulating conservation targets and were given two points each. Crisis Ecoregions and Last of the Wild use indicators for ‘non-provision probabilities’ only and therefore receive no points for this criterion.

There is a clear ‘flexibility’ advantage for those approaches which generate quantitative values as a basis for selection choices, and those with a technically well defined way of combining the generated values. Additional or alternative information such as on costs can systematically be integrated into these approaches. This applies to six of the presented approaches which we give two points each (Crisis Ecoregions, Biodiversity Hotspots, Global 200 Ecoregions, Wilderness Areas, Frontier Forests and Last of the

Wild). Although Endemic Bird Areas provide a clear scoring technique, the combination of scores for biological importance and threat level lacks explicit weighting which complicates ranking and integration of costs and other variables. Megadiversity Countries could easily integrate indicators for cost and 'non-provision probability' into the applied step-wise selection approach. However, lack of clarity on use of some of the secondary criteria (e.g. higher level diversity) compromise its flexibility. We therefore deduced one point from the scores of both approaches. The flexibility of Centers for Plant Diversity is mainly curbed by its use of qualitative information for many of its criteria and therefore receives a score of zero.

Looking at the overall weighted scores, we identify the top approaches to be Biodiversity Hotspots, Crisis Ecoregions and Endemic Bird Areas. In this multi-criteria analysis Crisis Ecoregions distinguishes itself from Biodiversity Hotspots by not including biodiversity service indicators and having lower data requirements. Frontier Forests comes fourth. With its focus on forest ecosystems the approach may not be the preferred choice for an IPES scheme, however. At the bottom of the ranking list we find Last of the Wild and Megadiversity Countries. Although Last of the Wild scored high in three criteria, its main weaknesses are no focus on private land, no use of any biodiversity service criteria, and a measure which targets regions that are likely to provide services even in the absence of payments. Megadiversity Countries has weaknesses in most applied criteria (except the service indicator).

Considering the spatial dimensions of the top three approaches there is considerable overlap, particularly between Biodiversity Hotspots and Endemic Bird Areas (Brooks et al. 2006). The main differences are that Biodiversity Hotspots cover large parts of the

Mediterranean region which are entirely omitted by Endemic Bird Areas. It also extends further into South-East Asia than Endemic Bird Areas whereas it does not include New Guinea as a priority area. Compared to Endemic Bird Areas and Biodiversity Hotspots, Crisis Ecoregions covers additional areas such as large parts of the temperate regions of North America, Europe and Asia as well as the entire African Sudanian Savanna and the complete Indian Subcontinent. The global cost map by Naidoo and Iwamura (2007), however, indicates that the inclusion of costs would likely reduce current priorities in the temperate regions and India. Consideration of service indicators would further change its priority regions possibly bringing Crisis Ecoregions closer to the other two approaches. The extent to which priorities will change with the integration of costs will depend on the spatial heterogeneity of costs within the existing sets of priority areas. The map by Naidoo and Iwamura (2007) indicates, for example, that gross economic per hectare rents are higher in the Brazilian Atlantic forest than along the Andes but exact conclusions can only be made when all selection criteria are systematically integrated.

6. Conclusions

Targeting is an important component for increasing the efficiency of payments for biodiversity services. The main targeting criteria that should be incorporated into allocation mechanisms are service level, cost and the probability that services are not provided in the absence of payments. Further targeting criteria could include readiness and leakage. Biodiversity services on the global scale are likely to first be targeted at developing nations where high biodiversity concentrations with significant threat exposure can be conserved at a relatively low cost. Data availability is probably one of the main constraints of global targeting. A stepwise selection approach could partly overcome this problem. There already exist a number of first-step prioritization templates for biodiversity services. Using the framework of a multi-criteria analysis we identify Biodiversity Hotspots, Crisis Ecoregions and Endemic Bird Areas as those most suitable for an international PES scheme. The ranking is to be interpreted with care, however, given that criteria and scores may change as the concrete objectives of IPES are being defined. A major shortcoming of all presented global prioritization templates is the neglect of conservation costs. While much work has been done on the prioritization of areas based on biodiversity indicators and threat future research needs to increasingly focus on methodologies for the assessment of associated costs, the provision of global cost data and their integration into global priority assessments.

This paper presented some of the diversity of targeting options which are currently available and pointed to some of the issues that need to be considered in IPES targeting. An IPES targeting design will largely depend on the to-be-defined objectives of IPES and the interests of the involved stakeholders in particular. We hope this paper provides information and arguments which can serve IPES stakeholders as a starting point to develop a targeting scheme that adequately supports international payments for biodiversity services.

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Tables

Table 1 Global targeting approaches by criteria and integration technique

Approach	**Target**	**Service indicators**	**Cost indicators**	**Non-provision indicators**	**Integration**	**Remarks**
Crisis Ecoregions (Hoekstra et al. 2005)	Biodiversity and ecosystem services at highest threat	None	None	Conservation Risk Index (Ratio of percentage area converted to percentage area protected)	Not applicable	Study unit: Ecoregions by Olson et al. (2001)
Biodiversity Hotspots (Myers et al. 2000)	Biodiversity in regions with concentrated endemism and high 'non-provision probabilities'	Number of endemic plant species	None	Per cent area of pristine vegetation lost	Stepwise: first by service level, then 'non-provision probability' indicator	
Endemic Bird Areas (EBA), (Stattersfield et al. 1998)	Endemic plant and animal species	Restricted-range bird species (with ranges <50,000km^2)	None	'Red List' threat levels converted to a score which represents surviving probability	Not applicable	EBAs are areas in which the ranges of at least two restricted-range bird species overlap
Centers of Plant Diversity (WWF and IUCN 1994-1997)	Areas of high plant diversity	Species richness and endemism. Also: value to humans, habitat diversity, edaphic adaptations by species	None	"The site is threatened or under imminent threat of large-scale devastation" (not further specified)	Stepwise: first by main criteria, then others (including threat)	Selection not primarily technical but through expert workshops sometimes involving an "element of subjectivity"

Megadiversity Countries (Mittermeier et al. 1997)	Countries with highest levels of species diversity and endemism	Endemic higher plant species and non-fish vertebrates. Other criteria: diversity of species, higher levels, ecosystems, presence of marine ecosystems and tropical rain forests	None	None	Step-wise: by number of endemic plant species (>2% of global total), 2nd by endemic non-fish vertebrates (>1% of global total). Knock-out criterion: presence of marine ecosystem. High diversity (>2,000 non-fish vertebrates and >10,000 higher plants can compensate endemism criteria. 'Other criteria' used less systemically	
Global 200 Ecoregions (Olson and Dinerstein 2002)	Biodiversity in general representing at least one ecoregion for every biome within every biogeographic realm	Species richness, endemism, taxonomic uniqueness, ecological and evolutionary phenomena, global rarity, intactness	None	None	Individual indicators are weighted and summed to produce overall score	
Wilderness Areas (Mittermeier et al. 2003)	Biodiversity in huge undisturbed regions	Intactness (at least 70% of historical habitat extent intact)	None	Human population density <5/km^2	Step-wise: first indicator for 'non-provision probability' , then service indicator	Only areas larger than 1m ha selected
Frontier Forests (Bryant et al. 1997)	Large, relatively undisturbed, natural forests	Contiguity, size (supports wide ranging animal species and resilient to natural disasters), human influence, naturalness (indigenous	None	Ongoing or planned human activities (e.g. logging, conversion, mining), likely future human activities (e.g. due to presence of natural resources)	Simultaneous consideration of service indicators. 'Non-provision probability' indicators considered as second step criterion	

		populations)				
Last of the Wild (Sanderson et al. 2002)	Areas where conservation creates least conflict with humans. Objective to find regions for every biome within every biogeographic realm	None	None	Population density, land transformation, accessibility, electrical power	Normalized scores for each indicator summed and converted to values 1-100 to obtain Human Influence Index (HII)	

Table 2 Multi-Criteria-Analysis of Global Targeting Approaches

	Transparency	**Low Data Requirements**	**Focus private land**	**Use of 'non-provision probability'**	**Use of Service**	**Flexibility**	**SUM**
Maximum achievable points	2	2	2	2	2	2	12
Biodiversity Hotspots	2	1	2	2	2	2	11
Crisis Ecoregions	2	2	2	2	0	2	10
Endemic Bird Areas	2	2	1	2	2	1	10
Frontier Forests	0	0	2	2	2	2	8
Global 200 Ecoregions	2	1	1	0	2	2	8
Wilderness Areas	2	2	0	0	2	2	8
Centers of Plant Diversity	0	0	2	2	2	0	6
Megadiversity Countries	1	1	1	0	2	1	6
Last of the Wild	2	2	0	0	0	2	6
MEAN	1,4	1,2	1,2	1,1	1,6	1,6	8,1