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Internalizing Global Externalities from Biodiversity – Protected Areas and Multilateral Mechanisms of Transfer

by

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Abstract:

Biodiversity can sometimes only be preserved if natural habitats are excluded from human uses. Such protection measures generate positive externalities at the global scale. This holds especially for protection in developing countries that host great parts of global biodiversity. For internalization, financial resources are raised on a multilateral basis and transferred to the host countries. This paper reviews the rationale for protected areas and transfer payments and summarizes empirical data. The resources provided through multilateral mechanisms - even together with official bilateral aid and private spending - fall short of estimated needs for effective protected area systems in developing countries.

Keywords: Biodiversity, International Development Assistance, GEF, Land Use, Protected Areas

JEL classification: N5, O13, Q2, Q5, Q56, Q57

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Contents

1 Motivation

Valuable endowments of biodiversity can sometimes be preserved only if natural areas that serve as carriers of these endowments are protected and excluded from most human uses. Such an exclusion of uses is enacted when even sound human inferences in natural areas lead to irreversible damages on species' living conditions and the functioning of ecosystems. Since the occurrences of such undesirable impacts may not result from interferences in a proportional way but happen in an unforeseeable way due to biological thresholds, command and control-regulation of land uses is often preferred to some 'price-based' regulation (Barbier 2000, Perrings and Lovett 2000, Perrings and Pearce 1994).

This paper is about land use regulations. More precisely, it analyzes how an efficient allocation of natural areas between human uses and protection thereof can be organized. Given the wide spectrum of issues that the management of natural areas as protected areas involves (Munashinge and Mc Neely 1994), the following analysis is confined to the international aspects of managing protected areas. In this context, the issue consists of internalizing positive cross-border spillovers from protecting biodiversity when biodiverse ecosystems are withheld from land development. To resolve this issue a regime for international coordination has been implemented which we will analyze in more detail here.

The paper proceeds in the following way. Chapter 2 enfolds the *theoretical* basis for using protected areas as an instrument of biodiversity policy and highlights the international dimension thereof. When assuming that cross-border spillovers at the global level generally show a unilateral direction, internalization demands for transfers from countries with relative small biodiversity endowments to countries with relatively large endowments. In Chapter 3, the theoretical findings are translated into an *empirical* analysis of financial resources provided through multilateral mechanisms of transfer and the associated official bilateral transfers. Given the figures on international aid, Chapter 4 reviews the data on the actual extent of protected area systems as well as the financial needs for the management of a global network of protected areas. Chapter 5 summarizes and gives an outlook on further research directions.

2 The Economics of Protected Area Policy

This chapter summarizes the conceptual basis for using protected areas as an instrument of biodiversity policy. Starting from a generic perspective, the further description puts special emphasis on the international dimension of protected areas. In this regard, cross-border externalities among sovereign countries and international transfer arrangements for the internalization are studied. The chapter concludes with a brief theoretical analysis of incentive problems that may occur in the context of such arrangements.

The protection of natural areas is typically connected with restricting particular land uses that counteract with biodiversity conservation[1]. In this sense, the establishment of protected areas can be perceived as a problem of allocating land between different uses. Natural areas are typically characterized by a wide range of possible land uses (including the complete withdrawal from any extractive uses). In general, the type of land use influences the endowments of

[1] For the purpose of our study, biodiversity is defined as endowments of biological resources that are characterized by a high degree of diversity on the ecosystem level, among species and/or among genes.

biodiversity respectively its compositions within a considered area. Depending on the land uses and the resulting characteristics of biodiversity, natural (and modified) areas provide a bundle of tangible or intangible ecosystems services, which generate multiple benefits to human life.

The allocation of areas of land to specific types of land use results from the decisions of the owners of land titles. In practice, enforcing property rights of (natural) areas – especially in biodiversity-rich regions in developing countries – represents a serious problem and is often regarded as a major cause for a degradation of biodiversity endowments. However, for our purpose, we abstract from this problem and assume that land property rights are properly defined and perfectly enforceable[2].

Considering land uses at the local level, cross-border positive externalities from the biodiversity conservation at one site occur, if households or firms in other locations beyond the boundaries of the single land property, respectively beyond national borders benefit from ecosystem services that are generated by conservation. Global externalities of biodiversity conservation represent the special case of cross-border externalities when nearly everyone derives benefits in some way from conservation at the considered site.

Examples for cross-border externalities from biodiversity conservation at the local level, i.e. among of the individual (private) landowners, can be seen in some supportive ecosystem services which enhance private agricultural production, as it is often illustrated in the example of the beekeeper and the farmer. Cross-border externalities at the level of sovereign countries occur in

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[2] For more on the issue of imperfect land property titles, see the related literature like Mendelsohn (1994) or Angelsen (1999).

connection with reserves that expand across borders or with bilateral or multilateral waters, like lakes or rivers. In this case, the beneficiaries are often limited to a group of identifiable individuals or countries. An example for global positive externalities are the conservation of diversity among species and at the genetic level, which creates non use values like existence values or bequest values and option values (Sandler 1999).

The described ecosystem services are typically non-excludable and non-rival in their use (Sandler 1993). The individual agent who possesses the property right title on land as carrier of biodiversity endowments cannot exclude third persons from consuming the provided services. If he cannot appropriate the external benefits of ecosystem services that his lands generate, he consequently attaches an economic value to biodiversity endowments hosted on the lands that is relatively lower than the value from the societal perspective.

Given this wedge between individual and collective evaluation of biodiversity, a sub-optimal allocation of natural areas results whenever different types of land use are available for a considered natural area, and conservation (or alternatively some environmentally sound land use) is not the most profitable type from the view of the holder of land property rights[3].

To make a private or more generally decentralized allocation of natural areas coincide with a collectively optimal allocation, the positive externalities from biodiversity conservation have to be internalized. For this purpose, a mechanism is needed which makes the landowners whose actions generate the

[3] Non-excludability is crucial for the distorted allocation since for this reason, a beneficiary is able to circumvent any commitment to co-finance biodiversity conservation. If non-rivalry prevails at the same time, this implies that in principle the group of free riders can be quite large without diminishing the quality of the non-excludable ecosystem services.

external benefits consider them in their land use decisions. At the local level, internalization can be achieved if the recipients of external benefits provide compensations in return for maintaining particular services from biodiversity and the landowner agrees to abandon land uses that appear to harm biodiversity. If private actors fail to achieve an agreement or some external benefits are still not internalized, an optimal allocation within national territories can in principle be implemented through regulation by public authorities. When externalities from biodiversity conservation are often not confined to national territories but spread across national borders, internalization also matters at the international level. For this reason, sovereign countries cooperate to attain an allocation that at least represents an Pareto-improvement to the situation without internalization.

Externalities from Protected Areas at the International and Global Level

In contrast to the national level, individual countries are not subject to a supra-national authority that can force them to comply with a commonly agreed conservation objective and to implement specific measures within their national territory. Thus, internalization at the international or global level demands for cooperation of the involved sovereign countries where the individual country has to agree on collaboration on a voluntary basis (Heister 1997,e.g.).

Collaboration in this sense means that the countries reach an agreement on measures, which each party will implement and which serve the common objective to preserve biodiversity. From the perspective of a single country, it is beneficial to join in such an agreement or convention if it can expect that its welfare is improving when all contracting parties enforce the agreed measures relative to a situation without an agreement, i.e. the agreement represents leads to a Pareto improvement for the participating countries.

Generally, two types of conventions are conceivable. On the one hand, the individual countries commit themselves to enforce on-field measures within their own territory to attain a commonly agreed objective with regard to biodiversity conservation. In this context, every participating country enforces measures. Comparing the individual countries with each other, the absolute extent of protection efforts may vary among them; meaning that the overall burden of protection may be unevenly distributed between them. The fundamental feature of this type of convention is that each country carries the costs of the efforts in essence on its own. Let this type of convention be named *reciprocal convention*. On the other hand, protection measures can be carried out by only some of the participating countries while the others recover parts of the costs that occur to them. Thus, in contrast to the first type of convention, a country that enforces protection activities under an agreement is partly or completely compensated for the incurring protection costs (Endres 1995). Therefore, let this type of convention be named *compensation convention*.

The choice of the appropriate type of convention depends on the direction the externalities from protecting natural areas take. In the case of *multi-directional externalities*, each involved country depends on actions that the other countries enforce, but also each country itself causes externalities on others by its own actions. Typical examples for multi-directional externalities from environmental resources is the abatement of carbon dioxide in global climate policy or the reduction in the use of chlorofluorocarbon to prevent the depletion of the ozone layer. Accordingly, reciprocal conventions like the Kyoto protocol or the Montreal protocol where signatory countries commit themselves to reduce emissions within their territories are the prevailing instrument to internalize multi-directional externalities. By contrast, in the case of *unilateral externalities*, the countries that cause the externalities can be distinguished from the recipients of the externalities. Examples are bilateral negotiations between countries at the upstream and downstream side of rivers to reduce pollution. Here, the compensation type is applied (Bernauer 1996, Ströbele 1991).

Turning to biodiversity, externalities from protecting natural areas may be both unilateral as well as multi-directional. Considering the indirect use values from biodiversity, ecologists have often pointed at the interactions and repercussions among ecosystems in different regions are of importance for life-supporting functions at the local level (WBGU 2000:Ch.C). In this sense, protecting and stabilizing natural areas in regions with small endowments of biodiversity presumably also causes positive externalities on sensitive ecosystems in regions that host larger parts of global biodiversity. Nevertheless, it is generally agreed on that the flow of positive externalities provided by countries in richly endowed regions dominates the flow in the other direction.

This view is also expressed in Art. 20 of the Convention of Biological Diversity (CBD) where the economically developed countries that host comparatively few biodiversity endowments are committed to support protection efforts in the developing but richly endowed countries. In this regard, *compensation convention* has been determined as the predominant type of convention for preserving biodiversity at the global level.

Before further studying the implication thereof for global protected area policies, let us briefly discuss if there is a potential for an international *reciprocal convention* for protecting biodiversity. The subject of such a convention essentially represents an agreement on some country-specific quotas for the establishment and management of reserves that have to be fulfilled within the national territories[4] [5].

Fixed quotas on protected areas finally represent a regulatory approach for internalizing environmental externalities. However, as it is shown in the context of pollution abatement, price-based respectively quantity-based instruments like tradable permits are associated with larger cost savings than regulation by charges (Siebert 1998). Therefore, an obvious question is whether the commitments to preserve natural areas are better organized in a regime of tradable land use permits.

The conception of tradable land use rights or "transferable development rights" has been studied on a theoretical basis (Panayoutou 1994, Cervigni 1993). More recently, the implementation of this type of scheme at a regional scale has been investigated (Weber 2004, Böhm et al. 2003). There it is suggested that in spite of difficulties in defining a proper indicator for measuring and comparing biodiversity at different places, a scheme of tradeable land use rights at a

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[4] Another example on reciprocal relinguishment of land uses is the Antartica Treaty (Cullen 1994). In 1991, the 26 Consultative Parties of the 1959 *Antarctic Treaty* that actually claim use rights in the region agreed to ban mining for the period of the next 50 years.

[5] This type of convention has indeed been implemented in practice as it is shown by the directive *92/43/ECC on the conservation of natural habitats and of wild fauna and flora* (EU 2003), which is currently implemented by the member states of the European Community. This 'Habitats Directive' aims at establishing a transboundary network of protected ecosystems ('Natura 2000'). Considering a possible translation of this specific-type of convention from the continental level to the global level, it has to be remarked that the European countries are relatively homogenous in their biodiversity endowment, i.e. spillovers from protection are more likely multidirectional. Furthermore, they have already hand over some sovereign rights to supra-national EU institutions. Both factors actually support the application of a reciprocal convention; however, they do not prevail on the global level in the same way. Furthermore, there are several mechanisms of transfers within the EU for various purposes, which directly or indirectly address biodiversity conservation. Especially, the LIFE mechanism (*L'I*nstrument *F*inancier pour l'*E*nvironnement) within its

regional level may generate some welfare gains relative to administrative rules for designing protected habitats.

However, it is questionable whether such a regime can be applied at the transnational or even global level. This is mainly due to institutional and political impediments to a cross-border trade of land property titles (Swanson 1999). Besides the difficulties in implementing an effective regime in this regard, the developed countries would have to undertake extensive commitments within such a system, which they would have to fulfill predominately in the developing countries with biodiversity endowment in order to attain effective protection at the global level. As a consequence, such a scheme of tradable land use permits would de facto lead to a compensation convention.

In sum, the effective protection of natural areas at the global level demands for international coordination among sovereign countries that manifests in conventions on compensations, which the developed countries provide as transfers to the developing countries for enforcing conservation measures that contribute to the generation of ecosystem services of global importance[6].

LIFE-Nature programme aims at funding nature conservation projects in accordance with the 'Habitats Directive' (European Commission 2003).

[6] The notion of 'transfer' has been traditionally used in the context of public sector economics (for example, Boadway and Wildasin 1984). The provision of transfers is considered as a major government function aside the spending of resources on public goods. Transfers in general refer to the "transfer of resources from some individuals or organizations to others". Intergovernmental transfers which are studied here can be distinguished in two types – unconditional and conditional grants. The latter are given on the condition that the recipient complies with a certain behavior. The notion of 'compensation' in this respect describes the intended function of a conditional transfer: From the view of the potential donor, transfers are only necessary if he and the potential recipient disagree about the actual recipient's behavior. Then, the recipient agrees to change his behavior in the way the donors wants it only if at least his ex ante level of welfare is maintained.

Accordingly, the subject of such conventions – besides protected area measures in its conventional definition – has to be a mechanisms of transfer which facilitates a resource flow to the regions with abundant biodiversity endowments to induce protection in excess of the nationally optimal level. An agreement on both protected area measures and transfers has to be reached in negotiations between the involved states.

Incentives Problems in Contractual Arrangements for Protected Areas

As mentioned, any enforceable agreement by definition represents a welfare-improving institution, i.e. cooperation among sovereign countries yields an economic surplus at the global level relative to the non-cooperative situation without an agreement. Therefore, negotiations on the agreement have to deal with the surplus distribution among the involved countries.

When assuming that each country, no matter whether it provides transfers or conserves biodiversity endowments, maximizes its own utility and derives only limited utility from the well-being of the other countries, it then will also attempt to maximize its own payoff from cooperation. It can do so by behaving strategically in negotiations on transfer payments and conservation commitments.

As a consequence of strategic interactions, the cooperative outcome does not represent an efficient outcome with a globally optimal extent of conservation. An extent of optimal conservation would only be attained, it is not possible to increase the utility of any country by reallocating natural areas among different uses or by increasing the total amount of transfers without reducing the utility of any other country (Pareto optimum)[7].

In the following we frame strategic behavior by three categories of incentives problems. For this, we comprehend international negotiations among countries on protection measures in the following way: The two parties of negotiations are, on the one hand, the (developing) *host countries* which make sovereign decisions on the management of their natural resources. On the other hand, the (developed) *donor countries* try to influence these management decisions by providing transfers to the host countries conditional on the protection measures that they enforce.

Due to the fact that *many* countries receive global environmental externalities from preserving biodiversity as well as that externalities originate from protection in *many* host countries, negotiations can take place between individual donor and host countries or/and between several countries that cooperate or collude to a subgroup whose representatives negotiate on behalf of the group members. In practice, collaboration within one side of the negotiations can be observed for donor countries, as we will show in (3.1). In principle, it is also possible that there is a coordination or even collusion among host countries towards transfer payments from international donors. However, there is few evidence for such activities.

Figure 1 depicts the different interactions among the involved parties. There are interactions between the two groups but also among countries within one group.

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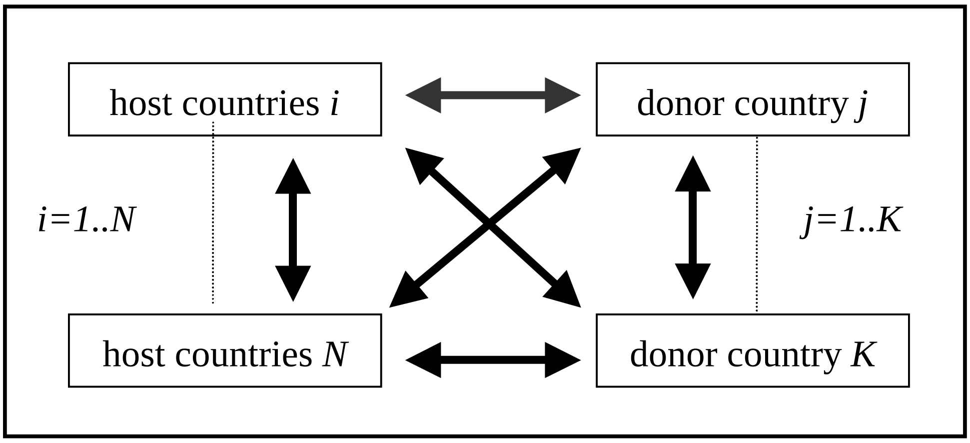
[7] In practice, a definition of efficiency relates to the selection of sites that should be protected. The rational decision-making on preserving some ecosystems in their pristine and undisturbed state and allowing extensive uses for others is methodologically a very complex and challenging task. A survey on the economic literature on this issue is given in Pfaff and Sanchez-Azofeifa (2004).

The structure of these interactions is not specific for an agreement on protected area measures. Moreover, it describes issues that is typical for multilateral co-ordination in general.

Figure 1: Interactions Among Actors in Multilateral Environmental Negotiations



The negotiation outcome can be affected by strategic behavior of host countries that arises because of the *irreversibility of biodiversity* loss. Ecosystem services whose potential decline would be irreversible may be represented by the preservation of rare species in sensitive ecosystems like tropical forests. Given these irreversibilities, strategic interactions take place against the background of repeated interactions among donor and host countries, i.e. to achieve effective long-term conservation, donor countries provide ongoing flows of transfers and renegotiate contractual arrangements with host countries over time. In this context, strategic behavior refers firstly to interactions among host countries and secondly to interactions between host and donor countries.

If several host countries compete for transfers in return for the conservation of some specific ecosystem services, the individual host country may have an incentive to undercut the competitors’ demand for compensations. It can do so by supplying conservation and demand for a compensation amount that is below its true costs. By this, some of its competitors eventually decide to abandon the protection of the relevant natural areas and thus to drop out of the group of future suppliers. By this “dumping” strategy, the countries that remain as suppliers of the considered ecosystem services are able to increase their bargaining power in the long run and to appropriate positive rents due to relatively higher future payments for comparatively low protection efforts (Stähler 1994,1992). When assuming that the donor countries provide a flow of transfers that is constant over time, the resulting path of conservation would be characterized by a relatively high level of protection in the medium term but a sub-optimally low level in the long run.

To inhibit or at least limit inefficiency in this regard, it is suggested to design a transfer scheme that limits the potential for strategic interactions among host countries. This can be done by fixing a maximum number of host countries with which contracts for conservation are concluded. This would render it impossible for any host country to attain any excessive bargaining power over time (Stähler 1994, 1992).

Considering the relationship between host countries and donor countries, the fact that biodiversity loss is irreversible can strengthen the bargaining position of the host countries. These countries may articulate the creditable threat that if donor countries do not provide a transfer amount that substantially exceeds the host countries’ costs of conservation, they will deliberately destroy their biodiversity endowments (“Burn the forest!”) (Mohr 1990). In other words, donor countries are requested to give up parts of their share of the cooperation surplus. If the donors do not react on this threat and destruction is made real, this would in effect decrease the external benefits they receive. If again donor and host countries repeatedly interact and the donor countries constantly refuse to redistribute the cooperation surplus, biodiversity endowments and thus the size of the surplus will shrink from period to period (Sandler 1993).

To avoid an inefficient allocation in this regard, it is important that donor countries are fully aware of their bargaining position in negotiations with the host countries. They should achieve an *early* agreement that establishes an adequate level of conservation and at the same time effects that destruction is no more a credible option (Sandler 1993). In this regard, several exogenous factors like the host countries' costs of depleting the resource stocks as well as potential domestic external benefits from conservation have to be considered since these factor determine whether destruction is credible anyway.

In addition to strategic interactions in the presence of irreversibility, the relationship between host countries and donor countries can be affected by *information problems* (Koelle 1995). Firstly, prior to an agreement, donors do not exactly know the true cost that accrue to the host countries when they will enforce an agreed level of conservation. Secondly, donors sometimes cannot observe whether the recipients of the transfers indeed fulfill the agreed measures or whether the final level of the conservation is also attributed to exogenous factors, like specific climatic or ecological incidences which have an impact conservation.

Given these information asymmetries, host countries have an incentive to overstate their actual costs of conservation and thereby increase their own payoff form cooperation. Since donor countries anticipate that host countries may show an opportunistic behavior, they provide a smaller amount of transfers compared to a situation where information asymmetries are absent. Given a lower amount of provided transfers, also a comparatively lower level of conservation will be contracted in the negotiation outcome.

To limit inefficiencies in the allocation of land uses, several instruments exist. These are signaling or screening activities as well as the use of incentive-compatible payment schemes. An examples is the establishment of competition for transfers among potential transfer recipients. With these instruments, donors try to induce the host countries to reveal his true costs and comply with the agreed activities. In addition, donors and host countries may overcome information asymmetries by collaborating in the process of planning and enforcing projects activities for which transfer payments are requested (Koelle 1995).

Finally, within the group of donor countries, there is the problem of *free riding*: The benefits which each countries receives from protection measures in host countries are represented by non-use values that are typically non-exclusive and non-rival in their consumption. Given the additive nature of benefits from co-financed conservation measures, the size of benefits the individual donor country receives is then determined by the total amount of transfers the donor community provides but only weakly related to the transfers it provides by itself.

Accordingly, the individual donor country has only few incentives to provide as many resources as it would correspond to its true evaluation of the received benefits. Moreover, it likely understates its true willingness to pay for the provision of the public ecosystem services in negotiations among donor countries. When in this respect every donor country relies on the contribution of the other donors, the total amount of resources that is collectively provided falls short of the amount that would be needed to establish a network of protected areas at a globally optimal extent (Wagner 2001, Barrett 1994).

Generally, free riding behavior of donor countries occurs in two different types. There are either countries that do not join any agreement on transfers in spite of the benefits they receive or countries that actually contribute money for transfers but to a too small extent in proportion to the benefits they receive. Hence, the first type (non-participation) is the extreme case of the more general second type of free riding.

The impact of free riding behavior can be confined to some extent if donors with a comparatively high willingness to pay for biodiversity conservation manage to link this issue to other issues which are of comparatively higher interest to the countries, which apparently free riding in financing conservation (Carraro and Siniscalco 1998). In this case, the latter countries would be willing to make higher contributions to biodiversity conservation in host countries than in a situation without issue linking. As far as issue linkages do not reduce the contribution of the countries with a high willingness to pay at the same time, the overall amount of transfers that are collectively provided in this way can be increased and thus a relatively higher level of conservation can be attained.

To sum up, economic theory on negotiations suggests that due to existing incentive constraints, international co-ordination that is manifested by conventions on compensations is likely to lead to a sub-optimal level of conservation, even if the impacts of such constraints can be confined to some extent by designing appropriate negotiation procedures and payment schemes.

3 The Current International Protected Area Policy

Taking the findings of the theoretical analysis of international protected area policies as a background, this chapter investigates how the current international protected area policy simultaneously takes place in a multilateral framework (3.2) and on a bilateral basis (3.3). The subsequent Chapter (4) is going to study in more detail what can be considered as the outcome of that policy. Furthermore, in (3.3.) the relationship between multilateral and official bilateral funding is explored. At the end of the chapter in (3.4), figures on international giving in the protected area policy by non-governmental organization is briefly reviewed.

Following the theory of negotiations from above, we focus on international agreements or conventions that at the same time arrange for (i) protected area measures in countries that host biodiversity endowments of global importance and for (ii) resource transfers to implement protection in excess of nationally optimal level. The analysis is carried out, first, with respect to *multilateral* arrangements – with many donor countries, typically represented by a single donor institution, and a single host country as a contract partner for protection and, second, with respect to *bilateral* arrangements – with a single donor country approaching a single host country. Besides classifying arrangements according to the number of parties involved in a transfer arrangement, it would be possible to distinguish them according to the affiliation of the donors in public sector and private sector spending for transfers. In the following, the focus will be on public spending, which mainly consists of inter-governmental transfers for biodiversity conservation. Private spending is briefly dealt with at the end of the chapter.

3.1 International Agreements on Protected Areas and Multilateral Mechanisms of Transfer

Generally, there are a number of *national* and *international* agreements, which directly deal with the protection of natural areas or indirectly address the allocation of land areas between different uses. Considering the whole range of them, they represent a developed and heterogenic system that provides a substantial set of different instruments and institutions. Focusing on the coordination at the international level, there is generally a considerable number of existing international agreements with provisions on protected areas. Most of these agreements originate from the time before the CBD and usually do not directly apply to the conservation of biodiversity in its entirety but aim at the conservation of specific species including their living conditions or directly at the conservation of endangered habitats. Examples are the 1968 *African Convention on the Conservation of Nature and Natural Resources*, the *1979 Bern Convention on the Conservation of European Wildlife and Natural Habitat* or the 1979 *Bonn Convention on the Conservation of Migratory Species of Wild Animals* (Matz 2003, Harrison 2002).

Most of the agreements have neither established a mechanism for the international transfers in return for the maintenance of environmental services nor made use of existing mechanisms to channel resources to the host countries, i.e. they do not represent compensation conventions. Of those agreements that provide for the establishment and the management of protected areas only the *Ramsar Convention on Wetlands of International Importance (RC),* the *World Heritage Convention (WHC)*, and the *CBD* have implemented such a mechanisms respectively have established a link to an existing mechanism (Matz 2003).

The RC and the WHC both follow a listing approach, i.e. protected areas under these agreements are registered as ‘Ramsar Wetland Sites’ respectively ‘World Heritage Sites’. Since the UNESCO Man and Biosphere Programme (MAB) – like the RC and the WHC – also addresses protected areas in a network of protected sites (‘World Network of Biosphere Reserves’), it is sometimes considered as a further international “regime” (Grant et al. 1998). Nevertheless, in contrast to the other networks, the MAB network is not governed by an international agreement and has not an own mechanism of transfer (Matz 2003, WBGU 2000:420). Furthermore, the scopes of the WHC and the RC often overlap with the MAB, i.e. many natural areas that are designated UNESCO Biosphere Reserves are also ‘Ramsar Wetland Sites’ or ‘World Heritage Sites’.

Even though the three regimes using the listing approach can vary in their institutional approaches to zoning and monitoring protected areas, the MAB regime shall not be considered in the further analysis, because it is not explicitly linked to a mechanisms of transfer.

For RC and WHC, such a mechanism consists of a treaty-specific environmental funds: the *Ramsar Small Grants Fund (SGF)* respectively the *World Heritage Fund (WHF)*. Both funds operate with a budget of relatively small size and regard themselves as having a catalytic role to enable countries to address relatively small-scale projects in order to make preparations to obtain funding for larger projects from other donors (Matz 2003).

The *Global Environment Facility (GEF)* is appointed as the CBD’s mechanism of transfer. As a funding institution, the GEF is not confined to the issue of biodiversity but it serves several international environmental agreements. It came into existence in 1991, i.e. before the CBD was signed. The aim of the GEF is basically to assist developing countries and transition countries in protecting the environment and in promoting environmentally sound resource uses and sustainable economic development. Generally, the GEF allocates grants to measures for the protection and conservation of biodiversity, which generate benefits at the global level. Referring to the CBD, the grants should compensate for the “agreed full incremental costs” of such projects (CBD Art.20 (2)).

In all three regimes, transfers represent as resource flows from donor to host countries which are both signatory parties of the referring agreement with its associated mechanism. Transfers are provided to host countries predominately on a conditional basis, i.e. they are earmarked to specific projects that address biodiversity conservation in some specific way.

The transfers are in effect intergovernmental grants, i.e. the recipient is typically a public authority in the host country that enforces protection measures. Transfers are given mainly in cash but can also occur “in kind”, i.e. as capital, technology or knowledge that is passed to the project participants. To describe total transfers, they are usually summarized in monetary terms, e.g. a transfer of knowledge is typically described by training costs.

The three mentioned mechanisms of transfer differ substantially in the total resources they provide. Table 1 gives some figures for two of the three mechanisms. Figures on the RSGF are taken from Ramsar (2003). It is suggested the RSGF is of relatively small scale since it has on average provided US$ 0.33 million per annum over the last decade.

Table 1: Provided Financial Resources by Mechanisms of Transfer by Year of Approval

Year	Ramsar Small Grant Fund nominal, in US$ M	Global Environment Facility focal area 'biodiversity', nominal, in US$ M
1991	0.139	162.775
1992	0.200	112.537
1993	0.318	31.416
1994	0.272	2.000
1995	0.293	85.397
1996	0.326	18.700
1997	0.734	193.463
1998	0.469	149.875
1999	0.400	191.797
2000	0.179	202.843
2001	0.197	205.376
2002	0.426	188.462

Note: Total grant volume of approved projects per year is assumed as annual grant volume. Grants under RC are originally given in CHF and converted in USD using exchange rates Thompson Datastream.

Source: GEF (2004), Ramsar (2003), own calculations.

The figures on the GEF originate from the online project database GEF (2004). They indicate that, after some volatility in the first years when the GEF itself and the link between the GEF and the CBD was established, the GEF fund has provided about US$ 200 million per annum. Furthermore, while the provided financial resources seem to be slightly but steadily increasing in the late nineties, this trend can not yet be confirmed for the last years.

The figures are calculated by adding up project grants by the year of the project's approval. For simplicity, these added up figures are considered as the annually provided financial resources. However, they do not indicate the actual payments in a single fiscal year. Since grants are usually disbursed over the entire project period and not transferred up front when the project starts – this is, however, implied by our procedure of summation – some distortions may arise. Nevertheless, since more detailed data on annual spending is not available at the aggregated level, the derived figures still represent a good approximation of the actual payments. Furthermore, the sum of the annual figures has to necessarily coincide with the actual total expenditures over the entire period and therefore the impact of any distortion is limited.

In contrast to the RC and the GEF, determining the resources provided by the WHF for protected (natural) areas is more difficult since the WHC does not only address natural sites but also cultural sites, i.e. man-made and urban sites which do not refer to biodiversity in a narrow sense. Considering the annual flows, Swanson (1999) quotes data on resources the WHF has made available in five two-years periods from 1983 to 1991. The provided financial resources on average amount to slightly more than US$ 2 million for the considered period. More recent data by WHC (2003) show that in 2000 US$ 2.3 million and in 2002 US$ 2.8 million have been provided for "international assistance". According to Spalding (2002), the total annual budget of the WHF is about US$ 3.5 million. Even when assuming that these resources are mostly directed to natural sites and thus have an explicit reference to biodiversity conservation, the total level of provided resources is still of relatively small scale in comparison to the means provided by the GEF.

To conclude, only a few of the currently existing international agreements that address protected areas measures simultaneously address a mechanism of transfer for internalizing global environmental externalities. Considering the existing conventions with respect to the provided transfers in quantitative terms, the CBD's mechanism plays the major role for promoting biodiversity conservation in a multilateral framework.

When connecting the identified figures on provided resources with observable outcome of the protected area policy, it is worth noting that not all of the resources are directed explicitly to protected areas measures since there are several instruments for conserving biodiversity besides protecting natural areas[8]. According to an analysis by the Worldbank (1995) on GEF biodiversity projects that have been approved during 1991 to 1995, 50% of GEF grants have been invested in the establishment and management of protected areas. Deke (2004b) finds out that of more than 600 projects approved during 1991 to 2003, 262 projects explicitly refer to protected areas. In this regard, the figures shown in the table represent an upper value of the total resources that the GEF mechanism has made available for the establishment and management of protected areas.

3.2 Official Bilateral Transfers and International Protected Area Policy

Besides providing resources within a multilateral framework, each country that is willing to contribute to biodiversity conservation in richly endowed countries has the opportunity to conclude contracts for protection and provide transfers on a bilateral basis.

A donor country may favor a bilateral arrangement if it believes that its priorities in conservation are better achieved than in a multilateral framework where the interests of other countries have to be taken into account.

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[8] For example, grants by the GEF have been provided for financing institutional capacity building in developing countries which however may indirectly contribute to the enforcement of the strict protection of ecosystems. In other projects, extensive human uses are not explicitly excluded but the implementation of environment-friendly management of natural areas are supported.

Dissent among donor countries about the conservation agenda may occur when the protection measures abroad do not only generate global public goods but also joint goods whose benefits accrue more to some countries than to others (Cornes and Sandler 1984). For example, a donor may favor projects that also establish eco-tourism capacities in host countries with a similar cultural background or a donor with a mature biotech industry may be interested in conservation measures that at the same time facilitate the preservation of genetic diversity. Bilateral transfers in this regard aim to finance the conservation of ecosystem services which from an inter-governmental perspective resemble more private goods than pure public goods.

In addition, an individual donor may have good contacts to a particular host country and is therefore able to contract biodiversity conservation to a price that is comparatively lower than the one that can be obtain in the multilateral framework. From the view of donor, funds for transfers are then more effectively used in a bilateral arrangement than in the multilateral one[9].

Otherwise bilateral arrangements may be less advantageous from the view of an individual donor country if host countries possess relative bargaining power that enables them to supply conservation activities at a high price or if information asymmetries with respect to the costs of conservation allows them to attract larger parts of the surplus from the financed protection activities. In contrast to this, it is assumed that in a multilateral framework, donors' interests

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[9] In this context, the reasons for bilateral financing may partly lay outside the environmental policy area: donor countries may give financial support for biodiversity conservation to honor political decisions made by the host country's government which are not directly connected to biodiversity, i.e. there is a somehow implicit linkage between environmental policy and other areas of international policy (cf. the discussion on 'issue linking' in the previous chapter).

can be bundled in a representative institution that can behave more like a monopsony and therefore make host countries to operate more closely at marginal cost than in a bilateral arrangement (Koelle 1995)[10].

Against this background, it is interesting to know how many resources for biodiversity conservation respectively protected areas have been provided on a bilateral basis in comparison to the resources provided by the multilateral mechanisms of transfer described before. For this, data on bilateral resource flows is analyzed.

Since bilateral financing is by definition decentralized and the provision of resources in this way takes several different forms like, e.g., in-kind transfers, debts-for-nature swaps or loans with a grant element, it is difficult to give a complete overview of bilaterally provided funds. Yet, to get an impression of the total funds that comes closest to actually provided means, we analyze data on bilateral financial flows provided in the OECD Creditor Reporting System[11]. This database contains information about bilateral outbound flows from 22 developed countries that are listed in the OECD Donor Aid Charts (DAC) plus flows from the European Development Fund (EDF). Recipients are developing countries and countries in transition.

The flows predominately represent grants like Official Development Assistance (ODA) or Official Aid (OA). As far as flows represent loans only their grant

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[10] Koelle (1995:151) considers the GEF as a monopsony representing the donor countries. By this, it is neglected that, in practice, the host countries can influence the decisions of the GEF as well since they have some voting rights in the GEF Council which approves the projects (Deke 2004a).

[11] In this regard, the focus is on transfers to developing countries. As described in footnote 5, there is a regional transfer mechanism in the European Union. Nature conservation projects within the EU in the context of the Nature 2000 network are supported by the LIFE-Nature fund with 300 € million from 2000 to 2004 (EC 2003).

element has been included in the figures. Since it is not possible to identify annual expenditures, figures on the single flows are again summed over the years in which the flows have been committed.

Depending on how to segregate the flows that address biodiversity conservation from other flows, the transfer amount presently ranges between US$ 200 to 900 million per annum.

Estimates for a high amount of total transfers are derived on the basis of "Rio Markers". These indicators are introduced by the OECD to mark transfers which refer to activities that are targeting the Rio Conventions. Flows that are marked in this way are either related to single environmental issues like biodiversity, climate change or desertification or to several of them. Table 2 indicates the figures for flows that address biodiversity only (first column) and for flows that address biodiversity in total, i.e. including flows that simultaneously refer to conservation and the other issues (second column).

Table 2: Financial Resources Provided Bilaterally as Official Transfer ("Rio Marker") (Grants by DAC countries; OECD-Rio Marker 'Biodiversity'; nominal in US$M, year of commitment)

Year	"biodiversity only" (Rio maker)	Total of Biodiversity (Rio maker)
1998	391.271	895.272
1999	466.189	783.659
2000	363.233	781.017

Source: OECD (2004); own calculations.

Since flows for 2001 and 2002 are apparently incompletely recorded, we only present for three years (markers are not assigned to flows before 1998). From these few figures, a trend in funding cannot be identified yet. Since the amount for biodiversity funds in total is about twice as high as the total amount for flows focused on biodiversity only, biodiversity conservation is obviously to a large extent addressed in connection with measures in climate change policies and in policies to fight desertification.

A look into the data shows that flows, which obtained a Biodiversity marker mainly fall into the OECD sectoral classifications “Agriculture”, “Fishery”, “Forestry”, “Water Supply & Sanitation”, but also “General Environmental Protection” and “Multi-sector Aid”. From these categories, it can be concluded that the marked flows also represent financial resources for activities, which do not aim at biodiversity conservation in the first place respectively, which address conservation in managed ecosystems or by implementing environmental policy institutions. Therefore, applying the derived total amounts to describe transfers for protected area management would overstate the amount that is actually provided for this purpose.

Furthermore, when comparing the figures on bilateral transfers with the GEF amounts in Table 1, it has to be noted that the GEF figures do not take into account the impact of projects in other focal areas on biodiversity conservation. Only recently, the GEF has begun to design and approve projects a with multi-focal scope.

The estimate for the low amount of total transfers is based on the OECD classification of flows by funding purpose. One explicit funding purpose in this regard is “Bio-diversity”. Table 3 depicts the amounts of annual transfers that have been bilaterally provided since 1989. To enable a comparison with the multilateral figures in Table 1, the figures are shown in nominal terms.

The flows in the first column show a similar path to one of the GEF funding: The annually provided resources have been increasing in the second half of the nineties but the margin of this increase has declined in recent years. Presently, the total transfer amount is about US$ 200 million.

In the second column, we add to annual figures for “bio-diversity” the amounts of flows on “site preservation”.[12]. It turns out that the transfers on “site preservation” are comparatively small and do not change the trend that is underlying the “bio-diversity” figures. For the same reasons as for the figures in Table 2, it cannot be ruled out, that these figures overstate the actual amount provided for protected area management.

The figures in the table shows that when starting in 1991, the year before the Rio Earth Summit, the bilateral funding decreases in the periods following the Summit – until the mid-nineties. From then on, the provided resources again increase and somehow converge to a level of US$ 200 million. For explaining the observed slow down in the increase of funds in recent years, two hypothesis can be formulated. On the one hand, it can be attributed to fluctuations in economic growth that cause fluctuations in public budgets which affect that a varying amount of funds could be made available in a specific year. On the other hand, other issues in international collaboration may emerge that also

[12] Similar to the payments by the WHF, flows in the category “site preservation” sometimes refer to the preservation of some human-made cultural or urban sites. Thus, when focusing on the amount that is made available for biodiversity conservation, there is probably some overstatement in these figures.

demand for bilateral funds and causes that international environmental issues is given a lower priority.

Table 3: Financial Resources Provided Bilaterally as Official Transfer (Grants by DAC countries; nom. in US$M, by funding purpose; year of commitment)

Year	"Biodiversity"	Biodiversity" and "Site preservation"
1989	4.450	4.730
1990	13.362	17.456
1991	82.843	88.138
1992	10.783	13.283
1993	54.620	63.084
1994	22.558	33.314
1995	97.848	108.674
1996	122.440	135.257
1997	71.601	80.014
1998	148.545	173.789
1999	183.086	246.361
2000	171.850	227.379
2001	185.981	204.404
2002	191.189	219.290

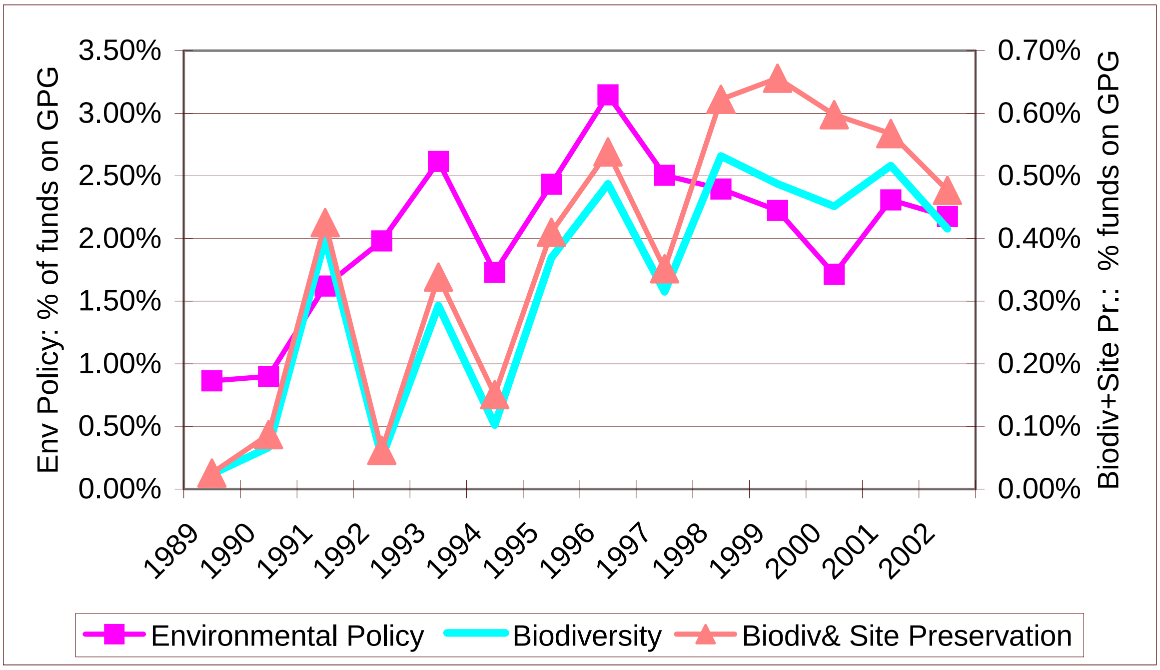
Source: OECD (2004); own calculations.

To test the two hypotheses, flows for the provision of "global public goods (GPG)" as defined by the OECD (2004) are studied (cf. Reisen et al. 2004)[13]. Figure 2 shows the flows for "bio-diversity" and "site preservation" as shares of

[13] "Global public goods" are defined by 40 categories including amongst other healthcare, family planning, energy issues, economic institutions (OECD 2004, see also Reisen et al. 2004).

the total financial resources that have been invested in GPG. During 1989 to 2000, the share for "Bio-diversity + Site Preservation" lies within a range of 0.03% to 066%. For illustration, "environmental policy" with a maximum share of 3.15% summarizes the other categories that refer to environmental protection.[14]

Figure 2: Share of Funds on "Bio-Diversity" and "Site-Preservation" Relative to Total Funds on "Global Public Goods"(only financial resources provided bilaterally by DAC countries)



Source: OECD (2004), own calculations.

The figure implies that environmental issues have attracted increasing attention in the early nineties with a peak in 1996. From then on, other issues have been given more priority relative to the environment. For "bio-diversity", the peak

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[14] The remaining categories refer to "General environmental protection", "Environmental policy and admin. mgmt.", "Biosphere protection", Flood prevention/control", "Environmental education/training" and "Environmental research" (OECD 2004).

was in 1999. The decline in the following four years corresponds to the slow-down in provided resources that is documented in Table 3.

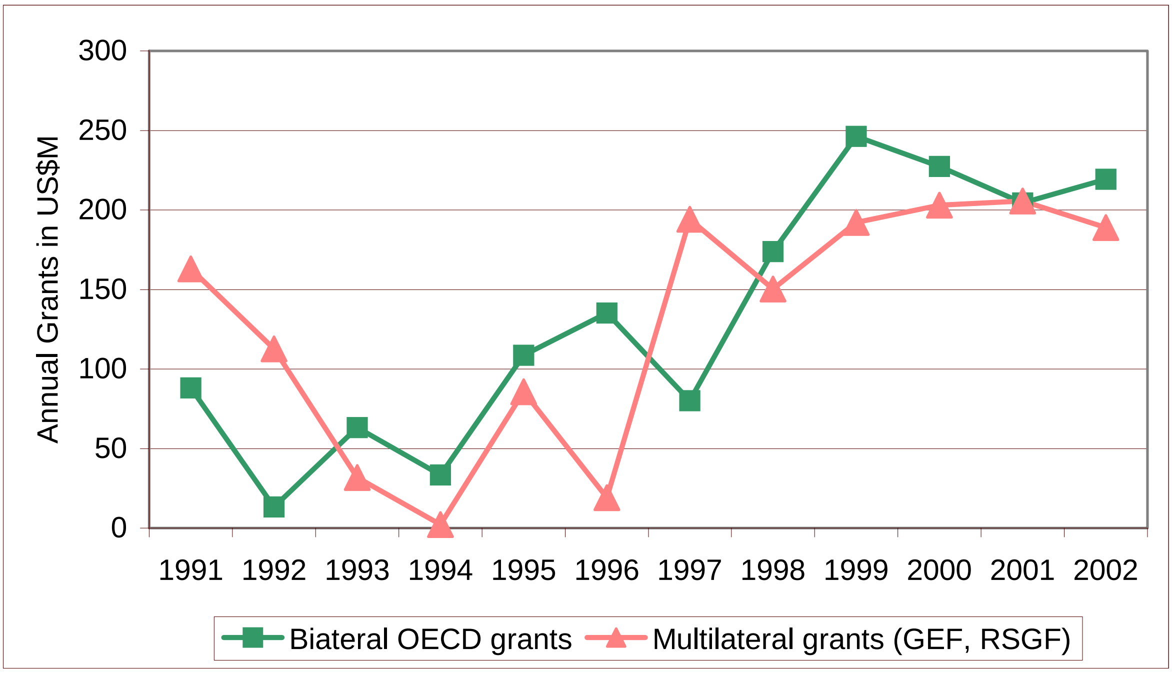
This result suggests that current expenditures on official transfers for biodiversity conservation are not subject to a slow down in economic growth that might be resolved in the near future (If this was the case, the share would have been rather constant and not decreasing over time). Moreover, the donors' willingness to pay for biodiversity conservation seems to be completely exploited.

The figures in presented Table 1 and 3 refer to nominal flows. For investigating the flows in real values, flows by multilateral mechanisms, i.e. grants by the GEF and RSGF, and bilateral flows for the purpose of "bio-diversity" and "site preservation" are described in Figure 3.

The figure reveals that the amounts for both types of transfers follows a somehow U-shaped course. Form this, two questions can be derived: First, have the financial resources provided in the second half of the nineties also increased in real values or can the observed increase in nominal values be attributed to increases in the price level? Second, are the financial resources that have been provided in previous years also larger in real values than the resources provided in 1991, the year before the Rio Summit?

Considering first, the annual changes of provided financial resources in nominal values. It can be shown that when the nominal flows are deflated with various price indices, there are – expect for 2001 – no noticeable reversals of the sign for these annual changes. Hence, prices changes do not have an overwhelming impact on the amount of provided resources. Furthermore, when using a GDP deflator for USA, it turns out that only since 1999 both the multilaterally and bilaterally provided resources are greater in real values than the ones provided in 1991.

Figure 3: Financial Resources for Biodiversity Conservation Provided Bilaterally by DAC Donors and Multilaterally by the GEF and the RSGF



Source: OECD (2004), GEF (2004),own calculations.

For investigating the effectiveness of these resources for purchasing of inputs in biodiversity conservation in host countries, the nominal flows are deflated can deflated by a price index for developing countries. By this, it is assumed that inputs for natural resources management are predominately purchased in the host countries. Depending on what type of deflator we use, it then turns out that, the real value of the resources provided in 1991 is mostly significantly higher than the one of the recently provided resources[15]. After all, the figures in real

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[15] To evaluate the purchasing power of resources transferred to the developing world it would be appropriate to use a producer price index. However, we have not found such a aggregated price index for the group of developing countries. We have therefore calculated

values for 1991 has to be qualifies since investments in biodiversity conservation have apparently been brought forward in the forefront of the Rio Earth Summit.

In sum, though it is difficult to identify the bilateral transfer that address activities in biodiversity conservation, the data implies that OECD countries currently provide about US$ 200 to 900 million per annum for activities in transition and developing countries. Since the funded activities address several measures, this figure is presumably overstates the funds that are provided for protected area policies in the recipient countries. The derived values at best represent an upper value for official bilateral funding in this respect.

Finally, these figures do not include donors outside the OECD. However, little is known whether oil exporting countries in the Mid East or countries in Latin America provide financial resources for environmental protection. It though seems reasonable to assume that official bilateral transfers by OECD countries represent the very large proportion of the amount of official bilateral transfers. The presented figures therefore give a good approximation of the resources that are transferred in this regard.

3.3 International Private Giving in Developed Countries and Protected Area Policy

The figures in (3.2) describe the amount of annual bilateral transfers that developed countries provide on an inter-governmental basis. However, in addition to official spending, there is also private spending that is used to preserve biodiverse ecosystems in the developing world. This is illustrated by

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with a consumer price index and acknowledge that the derived proportion of 5:1 likely overstates the actual proportion.

anecdotal evidence on some debt-for-nature swaps and other actions by non-governmental organizations (NGO) or non-profit programs by (multinational) firms in developed countries. However, since data on privately provided funds are not systematically recorded, it is difficult to determine the amount of annual spending worldwide in a reliable way. Results from a recent study on the international giving by philanthropic foundations may serve as an approximation in this regard (OECD 2003).

The study analyses the annual private spending in selected developed countries for the purpose of "development co-operation". It is found out that the major actors in this regard are US-based foundations. Figures for 2000 indicate that "conservation" activities including activities on "natural resources" and "wildlife" received 6.6% of international giving by US-based foundations. Considering a total amount of international giving of US$ 1.0 to 3.1 billion per year between 1994 to 2000, estimates for funds on conservation activities range between US$ 66 to 205 million. Contributions by European and Asian foundations are not documented in a comparative quality. Based on estimated figures, private European foundations provide US$ 350 million to activities outside Europe. Figures for international giving by Asian foundations cannot be derived in a similar way. (OECD 2003)[16].

When neglecting the Asian foundations and applying the relation between total international giving and giving for conservation activities that is observed for the US spending to grants by European foundations, the private spending on

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[16] Due to cultural and religious reasons, philanthropic giving in Asia is quite substantial. However, for the same reasons philanthropy is focuses more on local or national needs.

biodiversity conservation amounts very roughly to US$ 200 million per annum. In sum, even though data on private transfers is relatively poor in comparison to data on official transfers, the existing evidences suggest that the extent of private spending is substantial but below the extent of official spending.

4 A Global Network of Protected Areas – Outcome and Objectives of the International Biodiversity Policy

Having described the rationale for protecting natural areas and the current international policy in its legal and financial dimension, we are interested in how the outcome of such policy is manifests in natural areas presently put under protection. For this purpose, data on the actual extent of protected areas worldwide is summarized (4.1). Furthermore, recent studies on the extent and cost of an effective global network of protected areas are reviewed (4.2).

4.1 The Current Global System of Protected Areas

Several studies have described the status quo of the global system of protected areas and the system's development in recent decades (Green and Paine 1997, McNeely et 1994, e.g.). Recently, there have been efforts in compiling data on protected areas in a United Nations List of Protected Areas published by the World Conservation Union (IUCN) and the Conservation Monitoring Centre (WCMC) (Chape et al. 2003). Furthermore, a current summary of data is provided by the World Resources Institute (WRI) (WRI 2003).

When assessing the size of protected areas on a global scale, data problems occur due to a lack of reporting and differences in definitions of protected areas

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Overall, there has been little research on what Asian foundations are doing in developing countries (OECD 2003).

among countries. In this regard, the IUCN provides a meanwhile widely used classification of protected areas. In its basic version, this classification describes five categories of protected areas, which vary in the degree of exclusion from human uses (IUCN 1994). Protected areas of category I show the strictest protection from human interferences. In all subsequent categories, the exclusion is stepwise relaxed. Besides the IUCN core categories, further categories allowing for further or specific interferences are introduced. Especially, category VI, which allows for some sustainable resources extraction and ecosystem modification, has been taken into account in some studies on the size of protected area systems.

The United Nations List compilation makes use of the ICUN categories and aggregates national area protection systems in 15 regions are defined by the IUCN World Commission on Protected Areas (WCPA). The data we present in the following is provided by the WRI and relies to a large extent on the same data sources but uses an aggregation of 8 regions (WRI 2003). The figures are presented in Table 4.

The second and the third column of the table describe the extent of protected areas in absolute terms and its percentage share of the total land area of each region. The definition of a protection that is underlying these figures is very wide in the sense that the areas of category VI and areas that are not classified by the IUCN, i.e. areas with a comparatively low degree of use exclusion, are included. Since the study by the WRI also provides aggregated data on protected areas of categories I and II, i.e. Nature Reserves, Wilderness Areas, and National Parks, and protected areas of categories III to V, i.e. Natural Monuments, Species Management Areas and Protected Landscapes, the calculated percentage shares of these subsets are indicated in the fourth and the fifth column.

Table 4: Current Systems of Protected Areas by Regions

Region	Protected areas in national protection systems		Protected Areas Cat. I & II	Protected Areas Cat. III - V	Protected Areas as Part of Ramsar and WHC
	in hectares	in % of total land area	as % of total protected area	as % of total protected area	in hectares
Central Am.& Carib.	23 359 500	8.6%	22.1%	9.8%	5 236 000
South America	375 206 800	21.1%	17.7%	10.6%	20 676 000
Sub-Saharan Africa	264 389 600	10.9%	29.8%	24.0%	43 721 000
Mid East & N' Africa	118 797 200	10.2%	15.3%	15.2%	12 520 000
Asia (excl. M' East)	204 229 500	8.3%	43.6%	28.0%	8 275 000
North America	212 684 000	10.9%	47.0%	15.3%	34 634 000
Europe	180 720 900	8.4%	19.2%	52.7%	39 130 000
Oceania	66 095 400	7.7%	69.1%	11.6%	50 973 000
World[17]	1 457 674 000	10.8%	30.1%	22.4%	215 221 000

Source: WRI (2003, 2001); own calculations.

The figures show that the regional protected area systems vary in their structure. In most regions at least about 10% of the total terrestrial area is put under some kind of protection. Only in Oceania, Europe and Central America and the Caribbean less than 10% of the land surface are protected. The highest share is found for South America with 21.1%. As the figures next to it show, nearly three quarters of the protected land in this region is protected under a regime outside the IUCN categories and hence apparently subject to a relatively weaker degree of protection. A similar structure with a relatively high degree of human

[17]. The figures for the 'World' region are explicitly calculated in the study and do not precisely coincide with the sum over all regions indicated in the table (WRI 2003).

interferences in protected areas can be observed in the Middle East and North Africa. Here, only about 30% of the protected land falls into the categories I to V. In contrast, in Asia where only a comparatively low share of the land surface is protected, 44% of it is strictly protected and in another 28%, only relatively few human interferences are allowed. In the economically developed regions North America (Canada and the US) and Oceania (including Australia and New Zealand), there is strict protection for a large share of the designated land. In Europe, protected landscapes (Cat. III-V) dominate the protected area systems (53%)[18].

In sum, based on the figures for a wide definition of protected areas, 10.8% of the Earth's land surface is currently protected. However, if the protection of natural areas is confined to the first five IUCN categories, only about 5.7% or 764.95 million hectares are put under protection.

Considering the regional differences in the extent and the structure of the protected area systems, the possible reasons for these differences are multiple and difficult to verify on a general basis. Generally, it can be supposed that the extent of protection depends on the benefit-to-cost ratio of protecting natural land. In this context, scarcity of productive agricultural land or land for residential or commercial settlement can be regarded as major driving forces of the opportunity costs of protection. Combined with rapidly growing populations and strong food demands, this factor is probably dominant in North Africa and the Middle East and also in some parts of Latin America and East Asia (cf. Balmford et al. 2003).

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[18] For a detailed region-specific analysis of predominant types of protection, see also Chape et al. (2003).

In addition to the opportunity costs due to forgone land uses, the direct costs of managing protected areas have to be taken into account (Balmford et al. 2003, James 1999, cf. James et al. 1999a). In this regard, it is supposed that the staffing input that is necessary to assure effective monitoring and guarding tends to increase with rising population pressure. Empirical evidence though suggests that there are economies of scale in protected areas management, i.e. the necessary staffing input per hectare decreases with the size of the area (James 1999). Both factors may favor protection in less populated regions, which host some endowments of valuable biodiversity, like North America and Australia (Oceania).

The benefits of protecting natural sites are among other things the direct use values of biodiversity and can as well explain the extent of protection in some regions. For example, the use of wildlife in the tourism sector which can generate income in developing regions and thereby create incentives to protect natural areas as wildlife habitats. This may hold in particular for some politically stable countries in Latin America, Sub-Saharan Africa and Asia (cf. Norton-Griffiths and Southey 1995[19]).

Besides the use values, non-use values of biodiversity, e.g. benefits that are derived from the existence of a biologically diverse environment, occur in all regions. However, it is often supposed that a society's preference and willingness to pay for preserving the non use values of diversity is the higher the more income the people within a region receive. The resources that can be

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[19] The authors have studied the situation in Kenya and argue, however, that even though the current national protected area system generates substantial income, the current net pay off of that system for Kenya is negative. The authors conclude, that more financial resources should be provided on an international level to support and maintain Kenya's protected area system.

raised for protection within a region may partly directed to other regions; however, it is also reasonable to assume that a substantial proportion of these resources is spent on protecting the regional own regional biodiversity endowments (James et al. 1999b). Thus, relatively extensive systems of protected areas in the developed countries in North America and, to some extent, in (Western) Europe and Australia may benefit from the wealth that is generated on the remaining managed and modified areas of these regions.

With regard to the transfers in international biodiversity policy, the question is whether the data on the extent and structures of protected areas in the developing regions contains any information on the demand for transfers and on their efficient allocation among different regions. Considering the different national and regional protected areas systems, financial resources should be allocated according to ratio of external benefits to costs of protection in each systems. When assuming for simplicity that each unit of protected natural area generates fairly identical external benefits, then the differences in unit costs of protection would determine the allocation of transfers among the regions, i.e. regions with comparatively low unit costs receive a large part of the resources.

In this respect, it might be supposed that protection costs increase with the degree of exclusion of human uses, since this would increase the cost of monitoring and the demand for compensations for foregone uses. However, in contrast to this, it is often observed that natural areas, which are designed as strict reserves typically represent uninhabited areas. This in turn implies that opportunity costs of protecting such areas may be quite low – even in comparison to protected areas of categories III-V which allow for sustainable uses but may be connected with a relatively high demand for compensations (James et al. 1999b).

Consequently, a relationship between costs of protection and the different protection categories cannot be identified at the aggregated regional level. It is therefore – even under simplifying assumptions – not possible to derive reliable conclusions on an efficient allocation of transfers from the regional extent and structure of protected area systems.

A further question is whether natural areas that are assigned as protected areas by national governments are also part of the mentioned international agreements and are thus, eligible for possible funding by the associated transfer mechanisms. This does not hold for the GEF which does not pursue the approach of an explicit protected area network but for the two other regimes. Basically, protected areas become part of an international agreement if they are listed under a national protection system and are reported to the respective bodies or institutions of the agreement. An earlier study by the WRI (2001) contains aggregate data on areas assigned as (natural) heritage sites under the WHC or as Ramsar wetland sites. The sum of both figures is presented in the last column of Table 3. When we relate these figures to the ones of the first column, i.e. the total area that is protected, this gives some impression on how far these international agreements endorse the assignment of biodiverse natural areas as protected areas.

The shares calculated in the this way indicate that about 10% to 50% of protected areas in developing countries are listed under one of these two international agreements. This result can be interpreted in the way that there is apparently some acceptance of multilateral cooperation in protected area policies in developing countries. However, since otherwise substantial parts of the national systems are not integrated in these agreements, developing countries may have strong reasons on their own for designating natural areas as protected areas – independently of the chances to receive external transfers for biodiversity protection[20].

4.2 Resource Needs for the Management of the Global Protected Area System

Given the amounts of financial resources from multilateral and bilateral sources for a global protected area network as well as the figures on protected area systems at the regional and global level, a next step is to study to what extent resources match with the demand for an effective management of these systems and whether they suffice to assist for an expansion of these systems when assuming that such an expansion is necessary to ensure that biodiversity is effectively preserved.

In this context, it has to be determined what is meant with 'effectiveness'. Generally, it has to be considered that the establishment of protected areas is rather an instrument to preserve biodiversity than an objective on its own. Accordingly, recent policies have not explicitly addressed protected area targets but have focused more on components of biological diversity like species richness or the diversity among ecosystems. This can be illustrated by the "2010 target", which has been adopted at the 2002 World Summit on Sustainable Development in Johannesburg. The implementation of this target relies – among other measures – on the establishment and management of protected areas. This view has recently been expressed in the Kuala Lumpur ministerial declaration at the *7th meeting of the Conference of Parties to the CBD (COP7)* in February

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[20] A slight distorting impact occurs due to a double account of 20 sites that are part of the WHC and the RC at same time. (WRI 2001).

2004 as well as in the recommendations of the *5th IUCN World Parks Congress* in September 2003.

In this regard, the optimal extent of the global network of protected areas as well as its composition with respect to the degree of exclusion from human uses can hardly be determined on a global scale. In fact, the characteristics of an optimal network result in a bottom-up manner from the aggregation of local protection needs that are identified in various sites worldwide.

By contrast, in the past more explicit protection targets concerning protected areas have been formulated: In 1982, the IUCN released the *Bali Action Plan* whose recommendations for an expansion of protected area systems influenced the 1987 *Brundtland Commission Report* (Miller 1994, Sanjayan and Soulé 1997). This report recommended that, for effectively preserving biodiversity, the amount of protected areas has to increase three-fold relative to the amount at the time the report was published. Since at that time about 4% of the global land surface have been put under some status of protection, the recommendation has been loosely interpreted in the way that 10% to 12% of the global land surface is to be protected in some kind of way (Soulé and Sanjayan 1998)[21].

A 10% target is also underlying some studies that estimate the need of financial resources to implement protected area systems at a global scale. However, alternative ad hoc targets are also assumed in this context. In the following, the results of total (gross) costs of protection derived from four studies are reviewed. In particular, it is discussed to what extent the cost estimates can

[21] Recent recommendations by other institutions have followed the estimates of the Brundtland report like the German Advisory Council on Global Change (WBGU) who suggests that the appropriate conservation of multiple representative ecosystems on a global scale would result in a more or less strict protection of 10 to 20 % of the global land surface (WBGU 2000:413).

serve as approximations for the financial needs of a global protected areas network, or more precisely the needs in the developing countries (and countries in transition).

James et al. (1999b) suppose in their study that per se 10% of the land area in each of ten different continental regions (or a total of 1.6 billion hectares) is to be strictly protected. For the implementation of this target, 15% of the land surface would have to be put under protection. A protection to this extent would be associated with a total annual cost of US$ 27.6 billion. This figure (in 1996 US$) contains the costs of compensating local communities as landowners for their forgone revenues, the costs of optional land purchases as well as the costs of managing the existing and newly established protected areas. These costs figures are derived by extrapolation from with observed data on land values and management costs. Considering the detailed figures on regional demands, US$ 14.9 billion would accrue to the six regions that constitute the developing countries and countries in transition[22].

This latter amount apparently represents gross protection costs since no private goods like tourism services, which may be produced within protected areas and thereby generate some income to partly finance protection, are taken into account[23]. A potential caveat can be seen in that, similar to the Brundtland Commission Report, the study essentially relies on the assumption that a protection of one tenth of land area is "optimal" or the desirable level of

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[22] The study also describes the actual provided resources for biodiversity protection in each region. The expenditures in the six developing regions add up to US$ 695 million annually.

[23] One may argue that within in strictly protected areas, no private goods can be produced since nearly all human uses are excluded. However, note that according the IUCN Guidelines on Protected Area Management (IUCN 1994) tourism services are compatible with protected areas of Categories II and III which are sometimes included in a broad definition of strict protection.

conservation in each of the ecologically as well as economically diverse regions, whereas a justification is neither given on economic ground nor on ecological ground (Soulé and Sanjayan 1998).

A study by *Lewandrowski et al. (1999)* addresses the environmental set-aside at a global scale. However, this study does not implicitly refer to financial needs for compensation. The authors employ a global but regionally disaggregated computable general equilibrium (CGE) framework to calculate the total economic costs of setting aside land from agricultural uses. The CGE framework is a powerful tool for analyzing the economic adjustments that take place within an economy when the use of productive land is exogenously restricted, as it is done by protected area regulations. For this reason, the cost figures, in contrast to the figures in the other studies, also include secondary economic impacts due to changes in relative prices that arise when protected areas are established. In the study, eight economically defined regions are considered and a specific module is employed to describe different land productivities within each region. Then, scenarios for reducing 5%, 10% and 15% of the productive land endowments in each of different land productivity classes are studied. The annual costs of such protection strategies at a global scale are (in year 1990 US$) $ 45.5, $ 93.3 and $ 143.8 billion. More interestingly, the total costs for the three regions comprising the transition countries and developing countries are $ 16.1, $ 33.1 and $ 51.1 billion in the different scenarios.

This figures may be regarded as net costs since, due to the CGE modeling, revenues from private tourism goods are included in the simulation results. An open question is whether natural areas assigned as reserves in the past are considered in the scenarios or whether the model starts from an assumed level of zero protection. The study does not precisely answer this point. If the latter case holds, the underlying protection objectives would not refer to the total land surface of a region but to the currently managed agricultural area and thus consider protection targets that are much stricter than indicated.

Myers et al. (2000) advocating for the "hotspot-strategy" of biodiversity conservation present some figures on financial needs. This seminal study gives detailed advice on where natural areas with an exceptionally high level of species diversity are geographically located and how many financial resources would be necessary for their "safeguarding", i.e. it does not explicitly focus for the strict protection of the considered areas. More precisely, 25 biodiversity-rich biomes ("hotspots") are identified which together cover 210 million hectares (or 1.4% of the global land surface). For their protection, US$ 0.5 billion would be needed annually. This figure is based on the (ad hoc) assumption that the protection of a single hotspots would on average demand an annual amount of US$ 20 million, whereas no further explanations for the derivation of this figure is given in the study.

The assumption of equal costs of protection abstracts from the fact that the selected "hotspots" expand across natural areas which vary significantly in their size – from 0.2 to 35.6 million hectares – and are hosted in various countries which presumably show significant differences in land values and management costs which determine the costs of protection. Furthermore, the spatial expansion of the hotspots is defined by purely biological criteria – a biome is a hotspot, if it comprises 0.5% of global plant species diversity on its area, i.e. no economic considerations enter in this definition. From an economic point of view, the optimal size of a specific hotspot could be smaller or even larger, depending on the benefits and costs of protecting a marginal unit of land. Finally, the derived costs can be adjusted for our purposes by only considering costs for the 20 Hotspots in developing countries. The gross costs thereby reduce to 0.4 US$ billion.

Balmford (2003) updates the figures presented in James et al. (1999) and uses alternative estimates for land purchases. Furthermore costs of establishing and managing 10% of the sea as marine protected areas are included in the study. The costs of protection of terrestrial protected areas and marine protected areas are then roughly estimated with an annual amount of US$ 24.5 billion respectively of US$ 10 billion (both figures in 2000 US$). When using the above share of costs accruing to developing regions as identified by James et al. for extrapolation, the costs in these biodiversity-rich regions would amount to US$ 13.2 billion.

Table 5 summarizes the results from the studies. Though the figures from all studies represent rough estimates that should be handled with care, it is remarkable that the figures consistently indicate that total costs of protection rise with more extensive protection targets. If this fact is interpreted as a slight evidence of some reliability of the figures, it can be concluded that the actual annual demand for funds to finance the worldwide protection of natural areas lies within a range from US$ 0.5 to about US$ 150 billions.

When subtracting the costs that accrue to developed regions from the figures found in the studies, the costs of protection in developing regions lie in a range from US$ 0.4 billion to US$ 51.1 billion. In this context, James et al. (1999b) argue that today less than 12% of the worldwide annual expenditures on protection are made in developing countries. Furthermore if a global network of protected areas was expanded to an optimal extent and effectively managed, more than 54% of total costs would accrue to developing countries.

Table 5: Annual Costs for Managing a Protected Systems Worldwide and in Developing Countries

		Annually needed financial resources in billion US$; nominal		
Study	Protection Target	Worldwide	Developing Countries (DC)	
James et al (1999b)	15% of land surface in each of 10 regions (10% strictly protected)	27.5	14.9	Costs in 6 regions representing DC and transition countries
Lewandrowski et al. (1999)	Protect 5% [10%,15%] of (managed) land surface in each of 8 regions	45.4 [93.3, 143.8]	16.1 [33.1, 51.1]	Costs in 3 regions representing DC and transition countries
Myers et al. (2000)	1.4% of global land surface ("25 hotspots")	0.5	0.4	20 hotspots in DC
Balmford (2003)	15% of land surface in each of 10 regions + 30% of the sea surface	24.5 + 10.0	13.2	Extrapolated values; cf. James et al.

When using the cost estimates to determine the actual financial needs that have to be met by transfers from developed countries, two things have to be considered. First, cost estimates have to be adjusted to net costs by subtracting revenues from tourism (Norton-Griffiths and Southey 1995). Second, an efficient implementation of protected area policies requires that also the developing countries make financial contributions according to the incremental benefits they receive from protecting their own endowments in excess of the domestic optimum. Accordingly, the incremental benefits have to be deducted from the net costs of protection in developing countries to arrive at the actual demand for external financial resources.

Since especially domestic benefits from protected areas can be identified in a reliable way only on a case-by-case basis, the presented figures cannot be properly adjusted to describe the total need for international co-financing of protection. Therefore the derived costs can be regarded as an upper limit for the amount of transfers that should be provided internationally.

Taking into account the difficulties in approaching the actual demand for financial resources and the fact that the figures derived in the studies should be handled with care, the findings suggest that the resources that have been annually provided by developed countries (cf. Chapter 3) fall short off the required amount.

5 Concluding Remarks

Nationally enforced protected area policies do not only provide benefits to the enforcing country itself but also positive external benefits at the regional as well as global level. For this reason, some individual countries have committed themselves to nationally enforce conservation measures on a reciprocal basis, like in the European Community (EU 2003). However, since global biodiversity is very unevenly dispersed, this kind of reciprocity does not yield effective conservation at the global scale. Hence, (economically developed) countries that are endowed with less biodiversity endowments have agreed to support protection measures in (developing) countries with large biodiversity endowments by transfers payments.

This paper has given an overview on international agreements on protected areas and the associated transfer mechanisms both from a theoretical and an empirical viewpoint. The evident question that follows from the empirical analysis is whether the existing regime of multilateral mechanisms of transfer and the parallel bilateral transfers works properly to effectively preserve biodiversity at the global scale. It turns out that “effectiveness” in this regard is most essentially determined at the local level and it is therefore difficult to reliably define “effectiveness” at a global scale. Besides the problems in defining a common target with regard to protected areas, it is also difficult to precisely answer to what extent such a target is fulfilled since the necessary empirical data is either incomplete or subject to different terminologies.

In spite of these impediments, it is possible to derive some figures on financial resources that have been transferred in this context and on the demand for resources. The described stream of transfers consists of flows which are usually provided on a basis of proposed project activities. Since such activities do not only include protected area measures but also refer to other protection measures, the presented figures have to be regarded as an upper limit of transfers for protecting natural areas abroad.

These figures actually confirm that assisting developing countries in their protected area policies is recognized as important and is increasingly addressed in global biodiversity policy. This is manifested by an increase in official transfers by the GEF as predominant multilateral mechanism and by bilateral donors over the past decade whereby, however, this increase has not continued in recent years. When considering the three sources of transfers in developed countries, i.e. official multilateral mechanisms, official bilateral aid and private giving, the findings suggest that depending on its definition, bilateral transfers represent the largest flow followed by multilateral transfers. Private giving as is currently recorded is smaller than the official bilateral and multilateral spending by governmental institutions.

When believing that worldwide annual investments in protection natural areas are significantly below a level that would enable an effective management of global biodiversity by protected areas as it is argued by James et al. (1999b), the question is whether the reason for this has to be attributed to the fact that developing countries manage their own biological resources unsustainably or that the flow of transfers from developed country is significantly below the required level (Swanson 1999). Based on the findings of the theoretical analysis, i.e. the potential free-riding problems among donors, this second issue cannot be rejected.

This result brings up the question of how the current (multilateral) regime of transfers can be improved and which instruments or institutions can lead to an allocation that would be welfare-improving in comparison to the status quo. A further analysis of such a question may require a different level of investigation: So far the study on needed and provided resources that has been presented here is performed at a highly aggregated level. For improving the analysis of transfers for biodiversity protection, there is a need for specific information on protection measures on smaller scales (cf. Balmford et al. 2003), like on the size of protected areas that are addressed by transfers, on land ownership and potential land use rights in these protected areas, on the division of domestic baseline financing and incremental co-financing by external sources as well as on the sustainability of provided financing. An empirical analysis of specific protected area projects with respect to these aspects could be promising. In this context, it could be reasonable to focus on projects assisted by the GEF, as it is the dominant multilateral mechanism of transfer.

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