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Environmental Policy Instruments for Conserving Global Biodiversity

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Environmental Policy Instruments for Conserving Global Biodiversity

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Preface

If you can believe, all things are possible to him who believes.
—Mark 9:23

The current, unprecedented loss of global biodiversity as a result of anthropogenic interference in the world's ecosystems is affecting human well-being across the globe with increasing severity. It therefore represents a major challenge in international environmental policymaking. With the Convention on Biological Diversity (CBD), the community of states has recognized the increasing importance of preserving biodiversity.

Given the extensive context of biodiversity loss and preservation, this study focuses on two issues, which are at the center of the public discussion regarding the objectives of conservation and the sustainable use of biodiversity, and that are addressed by specific policy instruments. The first issue is the regulation of cross-border trade in genetic information and genetic resources. Here, the question is whether the commercial use of genetic information derived from biodiversity can create incentives for its preservation. The second issue involves the conservation of biodiversity through the protection of ecologically valuable ecosystems from human use. Here, the question is how the protection of these natural areas and the consequent restriction of destructive human use can be organized effectively on an international level.

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Kiel, September 2007 Oliver Deke

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α	proportion of preserved area
β_c	(gross) profit of a marginal unit of converted land
β_p	profit of a marginal unit of preserved land
γ	parameter describing the "production" of genetic resources
δ	discount rate
λ	scaling parameter (when calculating the value of a marginal species)
κ	scaling parameter (when calculating the average value of a species)
η	share of q newly approved drugs based upon natural products
ρ	parameter describing the agricultural production
π_c	profits in converted area
π_p	profits in preserved area
A	bioprospecting/natural area
c	unit cost of testing
$c(\ )$	conversion costs
$\hat{c}(\ )$	unit cost of conversion
c_g	unit cost of collection genetic resources
c_y	unit cost of resource extraction

D	species density
$\hat{D}$	species density in a hotspot subregion
e	endemic plant species in a subregion as percent of species richness in the entire hotspot
$f(r)$	regeneration function of the natural resource
$h(r)$	"supply" function of genetic diversity
K	out-of-pocket cost for the development of a new drug
m	number of species in a collection (in simultaneous testing)
N	number of species on earth
n	number of species in a collection
p	success probability of a single genetic resource in R&D
p_A	probability that species hosted in A lead to a success in R&D
p_g	price per unit of genetic resources
p_y	price per unit of resource extraction
p_e, p_g, p_y	output prices
$p_{x_e}, p_{x_g}, p_{x_y}$	input prices
q_e	public-good-like ecosystem services
q_g	genetic resources
p_n	success probability of the nth species in an ordered collection
q_y	agricultural good
q	average number of annual drug approvals
R	revenues in case of the successful development of a new product
r_t	resource stock in period t

s	size of the subregion as a percentage of the entire hotspot
$v(n)$	value of the nth (marginal) species
$\hat{v}_n$	"incremental value"
V_n	value of continuing the search with the nth species
$V(n)$	value of a collection of n species/genetic resources
V_{ph}	value of a newly developed pharmaceutical, ph
x_e, x_g, x_y	input in production
y_t	resource extraction in period t
z	species-area elasticity

Chapter 4

γ	parameter describing the accounting of incremental domestic benefits
δ	discount factor
θ_i	Lindahl share of country i
λ	parameter describing the relative weights in a welfare function
ρ	Spearman rank correlation coefficient
η	Lagrangian multiplier
μ_i, γ_i	share of country i in the finance of the public good
ω_i	weight of country i's contribution to the public good provision (weighted sum technology)
a, b, d	parameter in examples with functional forms
B	benefit from conservation
B_i	gross benefit of country i

b_i	country i's burden carried in the public good provision
C	costs of conservation
c	marginal cost of conservation
c_i	country i's consumption
$E(t)$	natural resource extraction
G	total supply of the public good
g_i	country i's contribution to the public good provision
L	Lagrangian function
M_i	money transfer by country i
n	number of countries
NB_i	net benefit of country i
$Q(t)$	natural resource stock
q	conservation
$R(t)$	natural resource regeneration
S	incremental environmental protection
s, v	parameter in example with functional forms
t	period
$T(t)$	transfer
U_R	utility of the resource country
U_{ROW}	utility of the rest of the world
u_i	country i's utility
V	maximum value utility of the resource country
Y_i	country i's income
z_i	joint product yielding benefits for country i

List of Abbreviations

ABS	access to genetic resources and benefit-sharing
AnGR	animal genetic resources
Art.	Article
bil.	billion(s)
BRS	biotechnology related sales
CBD	Convention on Biological Diversity
CGE	computable general equilibrium
CGIAR	Consultative Group on International Agricultural Research
CRS	OECD Creditor Reporting System
CTE	Committee on Trade and Environment
DAC	Development Assistance Committee
e.g.	*exampli gratia* (for instance)
EDF	European Development Fund
EU	European Union
EKC	Environmental Kuznets Curve
FAO	Food and Agriculture Organization of the United Nations
GDP	gross domestic product
GEF	Global Environment Facility
GET	Global Environment Trust fund
GNI	gross national income
GURT	genetic use restriction technology
HYV	high yielding varieties
IARC	international agricultural research center
IBRD	International Bank of Reconstruction and Development
ICDP	integrated conservation and development project
IDA	International Development Association
i.e.	*id est* (that is to say)
IEA	international environmental agreement
IPR	intellectual property right
IT-PGRFA	International Treaty on PGRFA
ITQ	individual transferable quotas

IU	International Undertaking on Plant Genetic Resources
IUCN	International Union for Conservation of Nature and Natural Resources (alias World Conservation Union)
LIFE	L'Instrument Financier pour l'Environnement
LMMC	Like-Minded Megadiverse Countries
MAB	UNESCO Man and Biosphere Programme
MAT	mutually agreed terms
MDG	Millennium Development Goals
mil.	million(s)
MRC	Minimum Contribution Requirement
MTA	material transfer agreements
NARC	national agricultural research center
NGO	nongovernment organizations
NME	new molecular entity
ODA	Official Development Assistance
OECD	Organisation for Economic Co-operation and Development
OP	Operational Programs
PBR	Plant Breeders Right
PIC	prior informed consent
PGR	plant genetic resources
PGRFA	Plant Genetic Resources for Food and Agriculture
PPP	Purchasing Power Parities
R&D	research and development
RC	Ramsar Convention on Wetlands of International Importance
rDNA	recombined deoxyribonucleic acid
SDR	special drawing right
SGF	Ramsar Small Grants Fund
SMS	safe minimum standard
STAP	Science and Technical Advisory Panel
TDR	transferable development rights
TEV	total economic value
TRIPS	Agreement on Trade-Related Aspects of Intellectual Property
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
UPOV	International Union for the Protection of New Varieties of Plants
USA	United States of America

WCMC	World Conservation Monitoring Centre
WCPA	World Commission on Protected Areas
WHC	World Heritage Convention
WHF	World Heritage Fund
WIPO	World Intellectual Property Organization
WRI	World Resources Institute
w.r.t.	with regard to
WSSD	United Nations' World Summit on Sustainable Development
WTO	World Trade Organization
WWF	World Wildlife Fund

1 Aim and Scope of the Study

The world's ecosystems are presently undergoing extensive change. Figures on worldwide land use indicate that predominantly ecologically diverse natural areas are being transformed into homogenized agroecosystems. At the same time, red lists of threatened species have repeatedly been extended, accompanied by renewed estimates on accelerated rates of the gradual extinction of species. In addition, there are signs of an increasing decline in the world's genetic diversity. Altogether, these empirical facts are attracting public attention as evidence of the loss of global *biodiversity* that has occurred over the last 50 years and that has reached unprecedented dimensions.

Biodiversity, or synonymously biological diversity, is a fundamental property of the natural environment. It comprises the diversity of the environment's characteristics, which have resulted from the process of evolutionary development, as well as the relationships between them. In other words, biodiversity represents the diversity of life forms on earth.

Biodiversity plays a crucial role in providing a flow of *ecosystem services* that contribute to *human well-being*. In ecological science, an ecosystem is defined as a naturally occurring assemblage of living organisms interacting as a functional unit with each other and their environment. From an ecological and socioeconomic perspective, biodiversity is a local and global phenomenon, since it affects all ecosystems and people.

The degradation of ecosystem services resulting from the loss of biodiversity adversely affects human well-being. In general, such degradation induces people to adapt their economic behavior in order to avoid to some extent its adverse impact on their lives. Nevertheless, there is a wide consensus among scientists and policymakers that adaptation can be extremely costly, in particular if man-made technologies are not (sustainably) available as direct substitutes for degraded ecosystem services. In addition, given that the global loss of biodiversity is effectively irreversible, people lose the option of drawing upon ecosystem services that, at present, do not noticeably contribute to their well-being but that can become vital in the future. Against this background, mankind faces the need to address the causes of the loss of biodiversity and implement measures to halt this trend of change.

While there are several direct and indirect factors that drive biodiversity loss, these drivers of change frequently interact and can vary across political regions and ecological biomes, i.e., regional groups of distinctive plant and animal communities. Nevertheless, immediate *anthropogenic interference* in ecosystems is identified as a major factor in causing biodiversity loss. Considering a specific ecosystem, anthropogenic interference can lead to both the support of local biodiversity and its decimation. On a regional level, certain types of land use management cause an increase in biological diversity; however, the negative impact, i.e., the displacement and destruction of biodiversity, indisputably dominates the general trend of change in most regions, as well as on a global level.

As a consequence, changes in the human interaction with nature have to be considered and enforced. Such changes are not typically costless. Taking into account the benefits generated from ecosystem services, it has to be decided where and how much biodiversity should be preserved. Regarding the objective of preservation derived from such cost benefit considerations and the existing humanly devised institutions controlling environmental uses, the latter apparently fail to provide for the desirable level of maintained biodiversity.

From a socioeconomic perspective, this outcome is primarily attributed to *market failure*, i.e., the fact that the private cost of using the environment does not coincide with its social cost. This effectively favors those uses that are nonsustainable from an ecological point of view. From an alternative view, beneficiaries of nonexcludable ecosystem services frequently do not participate in its provision. This means private actors cannot appropriate parts of the values generated from their efforts in preservation. Both features cause the overuse of biodiversity-abundant ecosystems and/or the undersupply of investments in their conservation and sustainable use, leading to the degradation of ecosystem services and biodiversity as such.

When these forms of externalities occur within national borders only, governmental authorities typically have the power to enact and enforce *environmental policy instruments* that enable the internalization of external costs and ideally support the desired level of preservation. Regarding cross-border or global externalities, the situation becomes more complex, since countries are sovereign in their decision to use the environmental resources residing on their territory. No supranational authority can force them to consider the impact of preserving their own resources on other countries' welfare. Internalization can only be attained by *cooperation among countries*.

In practice, cross-border externalities from conserving biodiversity endowments predominately originate from developing countries that have a fundamental interest in using their natural resources to foster their economic development or simply to feed their rapidly increasing population. Recipients of ecosystem services are developed countries, which only host a minor part of global biodiversity. Nevertheless, they rely on the availability of a large gene pool and attach a comparatively high value to the preservation of wild flora and fauna elsewhere around the globe.

Although these stylized, opposing positions greatly simplify reality, they roughly represent the political background to recent conflicts over the conservation and sustainable use of global biodiversity. Over the last two decades, biodiversity has become an important issue in the international environmental and resource policy arena. Efforts in this arena have resulted in the signing and ratification of the *Convention of Biological Diversity* (CBD).

With the CBD, the community of states, currently represented by 188 countries as parties to the agreement, recognizes the increasing importance of preserving biological diversity. The *objectives* of the CBD comprise the conservation of biodiversity, the sustainable use of its components and the fair and equitable sharing of benefits from their utilization, particularly with respect to genetic resources. Due to its integrated approach, the CBD is embedded in the context of complex economic, legal, and political questions. It is also intimately connected to the commercial uses of biodiversity and technological progress in this respect.

Regarding the objectives of the CBD, there exists a wide diversity of *policy instruments* on a local, national, and international level that lead to their support. The design of the appropriate policy mix depends upon several variables specific to the country, location, and prevailing threats to biodiversity.

Given the extensive context of biodiversity loss and preservation, my study focuses on two issues that are at the center of the public discussion regarding the objective of conservation and the sustainable use of biodiversity and that are addressed by specific policy instruments:

- The first issue is the regulation of *cross-border trade of genetic information and genetic resources*. Here, the question is whether the commercial use of genetic information derived from biodiversity can create incentives for its preservation.
- The second issue involves the conservation of biodiversity through the *protection of ecologically valuable ecosystems from human uses*. Here, the question is how the protection of these natural areas and the consequent restriction of destructive human use can be organized effectively on an international level.

With these two issues, I investigate in more detail the major characteristics of the allocation of biodiversity as an economic good, namely the use of market institutions as a means to support preservation and the parallel governmental interventions needed to regulate environmental use. In both concerns, my study focuses on the particularities of a policy on an international level.

Genetic resources are an essential input into research and development (R&D) in industries based upon biotechnological applications. Recent progress in these technologies, combined with prospects of future profits, has raised the expectations that the commercial use of genetic information can contribute to the preservation of biodiversity. More specifically, by laying down a framework of property rights to biodiverse areas and the genetic resources hosted therein, the holder of the rights can receive sufficient revenues to forgo alternative, biodiversity-degrading land use and maintain natural areas as biodiversity habitats.

In practice, genetic information is used in different forms and different mechanisms exist in order to allocate biological resources as carriers of genetic information. Depending upon the different commercial uses in the various industrial sectors, genetic resources display specific properties so that the allocation of genetic resources differs substantially from that of other renewable natural resources. Mainly because of its properties as an information good, the transfer and use of genetic information takes place in two different institutional frameworks: on the one hand, there is an *administered allocation* primarily governed by actors in the public sector; on the other hand, there is a *market allocation* with profit-oriented actors. With regard to the market institution, it needs to be assessed how enabling indigenous landowners and local communities to participate in the commercialization of genetic information can support the preservation of natural areas. In this respect, the specific role of the *scarcity* of genetic resources needs to be reviewed.

The first part of my study analyzes the hypothesis that the commercialization of and trade in genetic resources can significantly contribute to the preservation of biodiversity (provided that the institutional environment is well-defined). The objective of this kind of market-based regulation is to endorse and support other policy instruments. However, it obviously cannot replace them in the near future. Therefore, there is an ongoing need for other policy instruments, particularly for land use regulations protecting specific natural areas from destructive human use.

In this respect, a number of national and international measures and agreements that address the *protection of natural areas* have been implemented over the past decades. Previously, the protection of certain areas was primarily assumed to be in the interest of the individual state. International agreements, in addition to national policies, served the arrangement of cross-border coordination. In contrast, more recent policy explicitly considers a network of protected areas on a global scale as a means to support the objectives of the CBD. Therefore, mechanisms are needed that can arrange for protection on behalf of the international community, particularly in cases where an individual state either has no self-interest in protection or does not want to readily relinquish the use of its natural areas.

To fulfill these requirements, the traditional approach to protected areas, with its conventional definition of use restriction, has to be enhanced by an international *mechanism of (financial) transfers* in order to compensate for forgone alternative uses, particularly in the developing countries. Examples are bilateral debt for nature swaps, the multilateral World Heritage Fund and the *Global Environment Facility (GEF)*. The latter is of particular importance, since it serves as the financial mechanism of the CBD.

I structure the study as follows: Chapter 2 introduces the issues of global biodiversity. I describe the relationship between biodiversity conservation and human well-being and contrast it with the current threats to biodiversity and its underlying anthropogenic causes. Then I recapitulate recent international policies that address global biodiversity loss. Based upon this, I briefly delineate the economic underpinnings of the human use of biodiversity and the policies aiming at its preservation. To set the stage for further analysis, the focus falls on problems of global biodiversity preservation and international and global policies.

Chapter 3 deals with genetic resources and the market-based approach to biodiversity conservation. Since market transactions demand private or at least exclusive property rights for the traded good, I give a characterization of genetic resources as economic goods and present the academic and political debate on property rights. Without providing an in-depth analysis of the topics, I present the conceptual background of property rights to tangible genetic material and intangible genetic information, and describe the key issues currently addressed in political fora. I continue with a description of the nexus between market supply and incentives for preserving natural habitats. Potential disturbances due to imperfections in the institutional environment, as well as ecological constraints, are named and briefly classified. My further analysis relies on conventional neoclassical assumptions and concentrates on incentives for biodiversity conservation induced by market revenues from the commercial use of genetic resources. In this respect, it is a matter of concern how revenues are channeled to the in situ providers, what market price they can obtain, and how this drives their decisions on land use management. For this purpose, I compile and interpret empirical data on transactions in genetic resources and their commercial use in different industrial sectors. Subsequently, I provide a more detailed theoretical analysis of their commercial use in the pharmaceutical industry. Values for the industry's willingness to pay for natural habitat preservation are derived from numerical simulation and are reviewed critically. Finally, I discuss how different assumptions concerning the technological, competitive, and ecological environment of the market for genetic resources can influence the incentives induced for private conservation and the general prospects of this market-based approach to biodiversity preservation.

Chapter 4 studies the preservation of biodiversity as an international and global public good. I first describe the properties of biodiversity and ecosystem services as public goods and highlight their relationship to protected areas. I summarize the economic foundations of protected area measures and the international dimension of such a policy, followed by an empirical summary of the current system of regulation and its effect. My analysis continues with an empirical analysis of the GEF, which currently serves as a major international transfer mechanism to support the developing world in the provision of biodiversity as a global public good. Following a general description of the GEF biodiversity project portfolio, the further analysis concentrates on the projects that address the establishment and management of protected areas. I address questions regarding the sustainable finance of protected areas. Furthermore, the financing of the individual projects according to the incremental cost principle is studied in more detail and general implications of the interplay between the cost-effective use of international transfers and the international equity issue are deduced. Finally, given the problem of free riding in collective arrangements on public good provision, I analyze the finance of the GEF both on a theoretical and an empirical basis.

In Chapter 5, I relate the major findings of my study to questions of evaluating environmental policy instruments, such as economic efficiency, (environmental) effectiveness, distributional effects, as well as administrative and political feasibility.

Regarding the underlying methodological approach of the study, I recognize that biodiversity represents a complex phenomenon whose investigation generally demands the tools of several research disciplines. With respect to the overall research agenda, the natural sciences are particularly considered the leading discipline. In this context, socioeconomics has its strengths in analyzing and improving the arrangement of sustainable and socially acceptable natural resource management. Since many research disciplines are involved, which all draw on the knowledge of the other disciplines, there is an increasing awareness of the need for *integrated interdisciplinary* biodiversity research.

The research work underlying my study is not completely integrated. I place the emphasis on the *economic discipline*. Knowledge from the natural sciences and political science is considered as far as necessary. The study analyzes economic questions on the use of environmental policy instruments in international biodiversity policy. It thereby contributes to the socioeconomic component of an integrated research agenda.

The natural sciences fundamentally distinguish between *terrestrial* and *marine* ecosystems, whereas freshwater or coastal ecosystems often play a connecting role between the two. Without going into detail, it is evident that the ecological interactions and functioning, as well as the nature of human use in representative ecosystems of the two kinds, can differ considerably. Against this background, I recognize the importance of the world's marine ecosystems and particularly the threats to the biodiversity hosted therein. To focus my investigation, I only study resource management in terrestrial ecosystems, i.e., the study primarily deals with problems of land use management.

Furthermore, in order to describe the complex interactions between the ecological and economic system, the terms "preservation," "conservation," and "protection" are used in the academic and political debate. *Preservation* implies that the environment is left in an unaltered condition, i.e., it remains unchanged. *Conservation*, in turn, refers to protection from loss or harm, whereas *protection* relates more narrowly to the action that keeps the environment safe from harm. From this perspective, conservation refers to the prevention of extinction, while preservation demands the safeguarding of the living conditions currently prevailing and is therefore the seemingly stricter concept. In the debate, the two terms, preservation and conservation, are frequently used as synonyms. Especially when the continuity of biodiversity can only be guaranteed if the natural habitat remains unchanged, the similarity between the terms becomes apparent. Since the demand for conservation and/or preservation depends upon circumstances that prevail in a specific environment and that cannot be generalized in the context of the issues investigated, I use the two terms as synonyms throughout the remainder of the study.

Finally, regarding the ethical foundations of my study, my way of reasoning implies that the study mainly relies on a moderate *anthropocentric* approach to the policy of biodiversity preservation. This considers biodiversity as a means of satisfying human needs. Humans ultimately attach value to biodiversity. Based upon this, decisions on optimal resource management determine the human ecosystem interference that maximizes the well-being of the resource owner under given constraints of ecological productivity and resilience. In contrast to this, an analysis of biodiversity policy may rely on the ideas of *biocentrism*, which assumes that biodiversity has a high intrinsic value and that, thus, preserving biodiversity will consequently generate a value of its own. Although the two concepts seem to oppose each other at a first glance, in practice they do not occur in their purist form. Elements of both concepts can be combined, as shown, for example, by the introduction of the precautionary principle and safe minimum standards in the actual biodiversity policy. Combinations of anthropocentric and biocentric elements are also considered in the context of the protected area policy in the second part of my study. Overall, the conservation and use of biodiversity is also linked to some far-reaching aspects, such as the rights of future generations or species other than man. I recognize that a policy formulation in this respect has to rely on ethical considerations rather than on economic analysis.

2 The Issues

In this chapter, I introduce major issues in the context of global biodiversity and set the stage for a more detailed analysis of selected instruments in international biodiversity policy. Section 2.1 presents stylized facts on biodiversity as a global resource, the changes it undergoes, and the causes thereof. Section 2.2 provides a summary of recent policies aimed at curbing the unreasonable loss of biodiversity. In Section 2.3, the economics of the human use of biodiversity and the policies aimed at its preservation are delineated. The focus of attention falls upon the problems of global biodiversity preservation, and related international or global policies.

2.1 Stylized Facts on Biodiversity

In this section, I first consider the existing definitions of biodiversity and concepts for its measurement. Then, I characterize ecosystem services generated by a biodiverse environment. Based upon this, I provide an overview of recent data on the stock of the global resource "biological diversity" and the changes in this stock. Finally, I describe the driving factors that threaten this resource with irrevocable degradation.

2.1.1 Definitions, Indicators, and Life Support Functions

There are several definitions of biodiversity in the ecological and socioeconomic sciences. Nevertheless, the definition in the Convention of Biological Diversity (CBD) has become the prevailing one in the political and scientific debate (MEA 2005b: 18).[1] According to Art. 2 of the CBD, biodiversity is defined as the "variability among living organisms from all sources . . . and the ecological complexes of which they are part." Biodiversity includes "diversity within species, between species and of ecosystems."

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1 Among others, it has been adopted by the recent international scientific work program of the Millennium Ecosystem Assessment (MEA 2005b: 18).

Considering this broad definition, multiple and complex aspects of an integrated ecological-economic system are reduced to and encompassed by a single term. This conceptualization is actually helpful in reinforcing public awareness of the issue, as well as in accompanying the process on conservation policy. To investigate and address problems relating to specific biological units at specific sites, biodiversity needs to be defined more precisely and more specifically (MEA 2005b: 18f.).

With the help of *indicators*, biodiversity is assessed on many biological levels—from diversity among (cultural) landscapes and ecosystems to variations on a genetic level. Since biodiversity includes multiple dimensions, a precise or unique assessment as a single-valued figure that, for example, could enable the ordering of biodiversity at different sites is impossible (Eiswerth and Haney 2001; Purvis and Hector 2000).

In spite of this limitation, *species richness* is frequently used as the common indicator of biological diversity in the public debate. On the one hand, species richness is easy to assess relative to other measures on other biological levels; on the other hand, relying upon the concept of species richness, i.e., the number of species, has several drawbacks. In particular, no information is provided on the variation within a considered set of species, meaning that the number and *distinctiveness* of biological attributes embodied in the species is not addressed. This aspect is of interest to economists, since distinctive attributes are related to economic scarcity and the values of biodiversity, i.e., the choice of freedom it provides. In addition, the concept of species richness does not include two further aspects that are important for ecologic research, namely the *abundance* (population) of individual species or the combination of variation and quantities in the spatial dimension, i.e., *distribution* (MEA 2005b: 19ff.).

In practice, ecological science uses several methods to assess species richness combined with one or two of its dimensions. For example, species richness and spatial distribution are addressed by the concepts of "within-area diversity" (α-diversity, γ-diversity) and "between-area diversity" (β-diversity). Species richness and abundance are combined by indices that use weighted population figures, such as the Shannon–Wiener diversity index (Armsworth et al. 2004; Baumgärtner 2002; Crist 2003; Maigan et al. 2003). Finally, distinctive attributes in a collection of species are considered by phylogenic diversity indices (Weitzman 1992, 1993, 1998; Nehring and Puppe 2002). The choice of the appropriate indicator depends upon the spatial scale considered, as well as the underlying research question.

Biodiversity, Ecosystem Services, and Life Support Functions

Biodiversity as an ecosystem characteristic is considered an important factor in several ecological processes.[2] In ecological literature, these processes are described in terms of productivity, stability (including resilience), nutrient dynamics, and susceptibility to invasion (McCann 2000; Tilman 1999, 2000).[3]

The ecological processes manifest themselves in various ecosystem services that can be described according to different classifications. The classification in the recent Millennium Ecosystem Assessment distinguishes four major classes.[4] A special role is assigned to (1) supporting ecosystem services, such as nutrient cycling, soil formation, or primary production. These services are not subject to direct human use but refer to the general functioning and reorganization of an ecosystem. In this respect, the supporting ecosystem services assist (2) provisioning, (3) regulating, and (4) cultural ecosystem services, which directly contribute to human well-being (MEA 2005a: vi, 7, 2005b: 8).

The provisioning services include (1) food (crops, livestock, capture fisheries, aquaculture, wild woods), (2) fiber (timber, cotton, hemp, silk, wood fuel), (3) genetic resources, (4) biochemicals, natural medicines, pharmaceuticals, and (5) water.

Regulating services comprise (1) air quality regulation, (2) erosion regulation, (3) water purification and waste treatment, (4) disease regulation, (5) pest regulation, (6) pollination, and (7) natural hazard regulation.

Finally, cultural services involve (1) spiritual and religious values, (2) aesthetic values and (3) recreation and ecotourism.

These ecosystem services define the environment as a *life support system* (Siebert 1981). They contribute to human well-being in various ways.[5] Using terms from economic theory, the impact of an environment with a large biodiversity manifests itself through several characteristics (Heal 2004):

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2 From the ecological perspective, other factors aside from diversity, such as species composition, disturbance regime, climate, and edaphic factors, also have an influence on the aforementioned ecological processes (Tilman 1999).

3 Susceptibility to invasion, or invasibility, refers to an ecosystem's property of being subject to invasion by exotic species that may dominate and push out species indigenous to the ecosystem. It is sometimes suggested that invasion is less likely to occur in diversity-rich ecosystems (Tilman 1999). The concept of resilience refers to the question whether and how rapidly an ecosystem regains its equilibrium after a perturbation has occurred (McCann 2000).

4 Alternative but similar categories of ecosystem services are provided in Costanza et al. (1997), Daily (1997), and de Groot et al. (2002).

5 The Millennium Ecosystem Assessment defines the contributions to human well-being as "basic material for good life," health, security and "good social relations." Overall, these contributions guarantee a choice of freedom and action (MEA 2005a: vi, 103ff., 2005b: 8).

- The increased *productivity* of natural resources hosted in terrestrial and aquatic ecosystems: more specifically, there is empirical evidence that a high degree of biological diversity in an ecosystem can, over a certain range, induce a higher level of biomass production.
- Reduced risks that result from dynamic ecological repercussions, such as the spread of pests or diseases, and that threaten production processes and human life. A biodiverse environment in this sense serves as *insurance* (Weitzman 2000; Heal et al. 2003).[6]
- The ongoing provision of *genetic knowledge* for (bio)technological research and development (R&D). This refers to the use of genetic information embodied in living organisms (Sedjo 1992; Swanson 1996).
- The continuing support of *ecosystem services* that contribute to the ecosystem's integrity and, therefore, are crucial for the provision of the other services. These services are virtually nonsubstitutable or demand costly man-made substitutes.

To characterize biodiversity-related goods and services in economic terms, it is useful to classify them with respect to (1) access and use rights and (2) use externalities. The access and use rights are predominately a social construct. They determine whether third parties are excluded from the consumption of goods or services. Excludability essentially depends upon the design of property rights. In several cases, it is not possible to enforce use exclusion for technological reasons. The use externalities refer to the fact that biodiversity resources are often rival in their consumption. As it happens, for some of the resources, just the opposite is true: consumption by one party does not diminish the quality or quantity that can be consumed by others.

Figure 1, taken from OECD (2003), provides a generic typology of goods and services generated by biodiverse ecosystems. The figure summarizes three characteristics, namely rivalry, excludability, and locality in a two-dimensional picture, and thereby it preassumes that the latter two are highly correlated.[7]

In practice, not all of the biodiversity goods and ecosystem services included in the two classifications described are provided in each individual ecosystem. Moreover, given the natural characteristics of an ecosystem and the human interference therein, some goods and services dominate output more than others. In the recent People and Ecosystems Report by the World Resource In-

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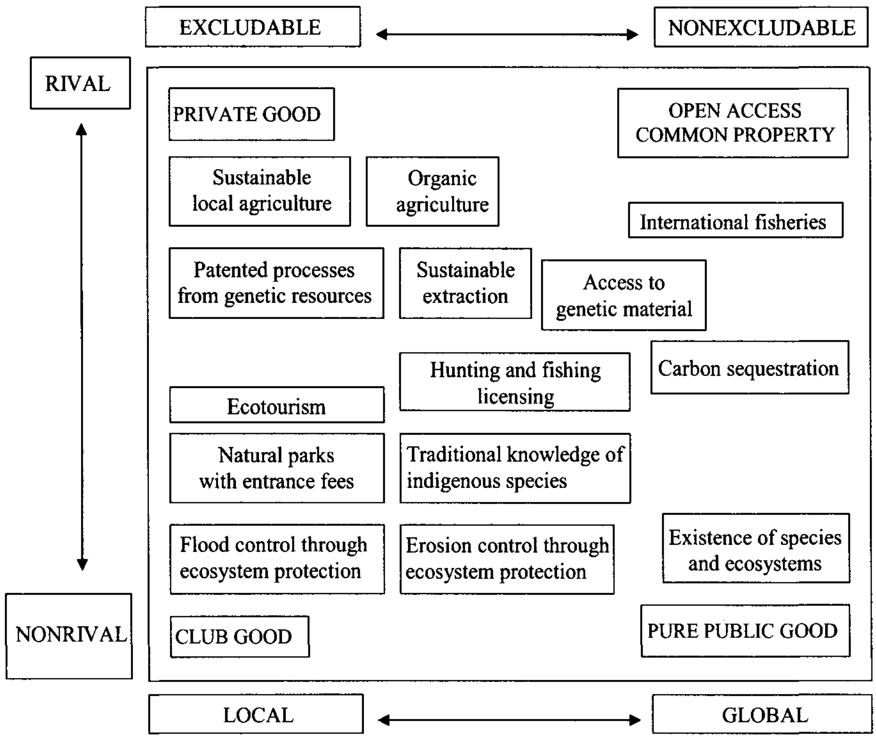
6 Alternatively, Folke et al. (1996) define the insurance function of biodiversity in the context of the stability/resilience of ecosystem conditions.

7 "Access to genetic material" is classified as a nonexcludable as opposed to excludable service. As I discuss in Chapter 3, the excludability of genetic resources depends greatly upon the various forms of the genetic material, its sectoral uses and the defined property rights regime.

Figure 1:
Biodiversity-Related Goods and Services



Source: OECD (2003: 30), see also OECD (2004: 26).

stitute (WRI), five forms of ecosystems are distinguished, namely agro-ecosystems, forest ecosystems, freshwater ecosystems, grassland ecosystems, and coastal ecosystems (WRI 2000). Each of these ecosystem forms is described by a representative bundle of goods and services.

While, in this respect, multiple goods and services result from the ecosystem management, it is shown that, in many cases, the provision of individual output components is not separable. That is to say, in the environment, there is a "bundling" or *joint provision* of specific goods and services with both private and/or public good properties (Heal 2003; Holm-Müller 1999).[8] In Section 3.2,

8 In managed ecosystems, particularly in agroecosystems in developed countries, this joint provision of ecosystem services is addressed by the concept of multifunctionality (Maier and Shobayashi 2001).

I pick up the idea of joint provision when assessing the impact of markets for the specific private good "genetic resources" on the preservation of public-good-like ecosystem services.

2.1.2 Biodiversity Loss and Its Causes

In order to assess strategies aimed at controlling the loss of biodiversity, information is required regarding the status of biodiversity as a resource and the change the resource stock is currently undergoing. In natural science, diversity can be assessed on different biological levels. The conventional classification distinguishes ecosystems, species, and genes. These different biological levels have to be combined with different spatial or geographical scales.

In addressing the need for information, there have recently been increasing efforts in data collection accompanied by technological advances in monitoring practices (e.g., satellite-based systems). However, the information gathered is still far from complete. Since there is no unique indicator of biodiversity, its assessment relies upon the use of multiple indicators. The choice of indicators largely depends upon the specific underlying question of biodiversity conservation and management (Eiswerth and Haney 2001).

Assessing Biodiversity as a Resource

Regarding the international or global level, which is in the focus of my study, biodiversity can be addressed on the level of *biomes*, which are broadly defined as habitat and vegetation types. The World Wildlife Fund (WWF) identifies 14 different biomes worldwide, e.g., mangroves or boreal forests/taiga, and describes their distribution around the globe (MEA 2005b: 2,9).

On the level of *species*, estimates of the global species richness across all classes range between total numbers of 5 to 30 million. Two things need to be mentioned here. First, only about 2 million species have been described formally, meaning that science relies upon a prolonging approximation when describing the total species richness (Plotkin 2000; Reid 1992).[9] Second, given the complex evolutionary processes, natural scientists have difficulties in providing a unique definition of species and accept a certain degree of inaccuracy in this respect (van Kooten and Bulte 2000: 272f.).

On the level of *genes*, diversity is reasonably not assessed across all species. Moreover, the attention in science is drawn to subsets of the species whose

9 Regarding the different species groups, some of them, like mammals or plants, are relatively well described, while, at the same time, there are significant gaps of knowledge for other groups, particularly microorganisms (MEA 2005b: 52).

genetic diversity determines the impact on human well-being. These are particularly plant species used for cultivation. Assessments in this area concentrate on distinctive material samples stored in gene banks (accessions). The relevant data in this context is reviewed in more detail in Section 3.3.2.3. A discussion of various biodiversity measures and their assessment is provided in Nunes et al. (2001).

Given the figures on worldwide biological diversity, patterns of its distribution around the globe can be identified. Habitats can be distinguished in terms of ecological and geographical parameters on the one hand and socioeconomic ones on the other. First, with regard to the biomes, it is shown that individual forms are distributed across different continents. Regarding species richness, investigations carried out by ecologists and biogeographers focus, inter alia, on the role of latitude as an explanatory factor. It indeed turns out that species richness is concentrated in terrestrial regions along the equator and that it decreases when moving towards the poles. Because of the relationship between latitudes and climatic zones, meteorological variables also show some significance (Gaston 2000).

Given these general characteristics, more narrowly defined regions can be identified that host exceptional species richness, i.e., regions that represent biodiversity "*hotspots*" (Myers et al. 2000). Again, most but not all of these hotspots are hosted in regions located on or close to the equator.[10] Similar to the species hot spots, "*centers of origin*/diversity", i.e., hot spots of wild genetic diversity, have been identified for major crop plants (FAO 1998: 20ff., 501ff.; Kloppenburg and Kleinman 1988). Although crop genetic centers of origin are not congruent with the species hot spots, they are also primarily located in equatorial or subtropical regions.

From a socioeconomic perspective, countries or economically integrated communities of countries in the tropical and subtropical regions predominately represent technologically less developed countries, which are currently faced with rapidly growing populations, implying an increasing need for food resources. In contrast to these *developing countries*, the *developed countries* in the more temperate climatic zones are endowed with a comparatively large stock of man-made capital but demonstrate rather slow-growing or even decreasing populations.

This picture is of course stylized and abstracts from differences within the groups of developing countries and developed countries. Furthermore, although climate and population variables may have an impact on a country's economic

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10 A related research question is whether there is a correlation ("covariance"/ "congruence") between richness in the various species categories. Studies lead to differentiated empirical findings in this regard. Generally, correlation is more strongly pronounced in the region along the equator (Gaston 2000; Myers et al. 2000).

development, I do not intend to identify or explain the underlying forces of economic growth here. The aim is to point out that, by combing the ecological and socioeconomic findings on a global level, there is an *asymmetry* of economic wealth and biodiversity (approximated by species richness).

This is illustrated by Figure 2, which includes data for 141 countries and uses GDP per capita as an indicator of wealth and species richness of higher plants for biodiversity. The hyperbolic, downward-sloping line supports the assumed asymmetric relationship between the two variables.[11]

In addition to these general trends, some countries can again be identified that host exceptional endowments of species within their national territory, indicating an exceptional richness in biodiversity. In the original definition, the 12 *mega-diverse countries* are Brazil, Colombia, Ecuador, Mexico, and Peru, in Latin America, the Democratic Republic of Congo, and Madagascar, in Africa, and, finally, China, India, Indonesia, Malaysia, and Australia, in the Asian and Pacific region (McNeely et al. 1990). In light of this definition, the political group the *Like-Minded Megadiverse Countries* (LMMC) was founded in 2002 to support and enforce common interests in the international policy arena. While Australia, being a developed country, is not included, six further countries, namely Bolivia, Costa Rica, Venezuela, Kenya, South Africa, and the Philippines have joined the LMMC. It is claimed that this group represents about 80 percent of the world's biodiversity and 45 percent of the world's population.

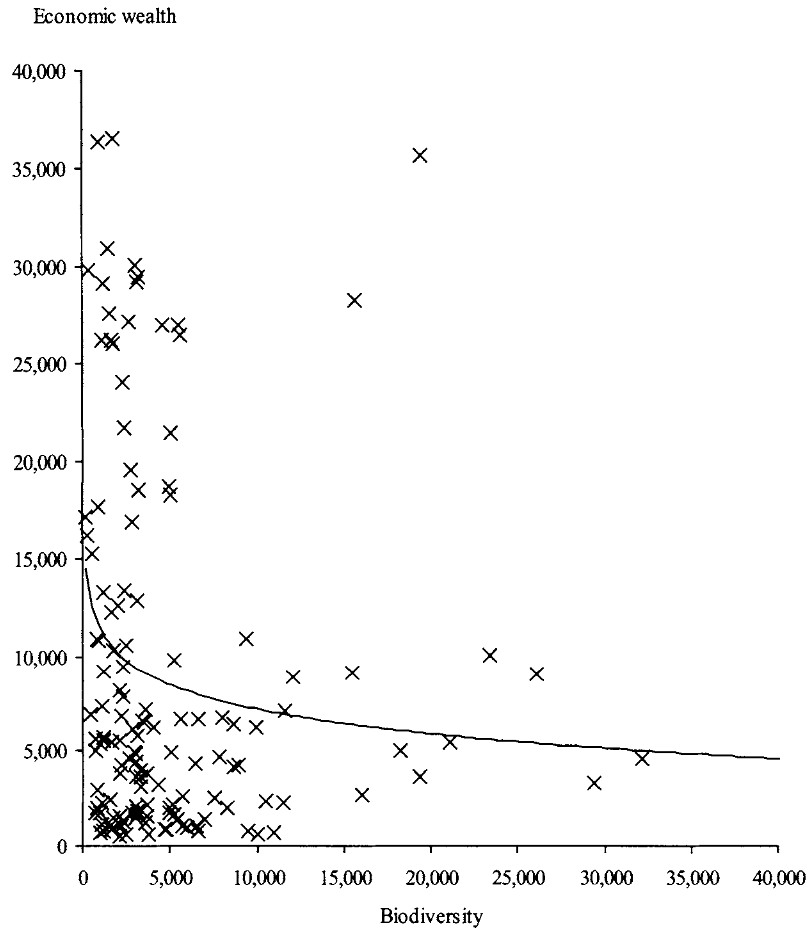
Biodiversity Loss: Changes in the Resource Stock

The size of the global resource stock of biodiversity has been subject to substantial variations in the historical and prehistorical past. The changes that are currently taking place are difficult to describe with absolute accuracy. However, the vast majority of scientists and policymakers interpret the existing evidence as signs of a drastic decline in global biodiversity of an unprecedented magnitude.

Empirical evidence of biodiversity loss is given in various figures on the different ecological and spatial scales. Focusing again on the global level, the notion that there has been a decline in *ecosystem* diversity is supported by observations on ecosystem modification from the second half of the 20th century: about 24 percent of the global terrestrial surface has been transformed from natural areas to cultivated areas. Regarding the different forms of biomes, the transformation has been greater in temperate grasslands, Mediterranean forests, tropical dry forests, temperate broadleaf forests, tropical grassland and flooded grasslands. On a global level, a deceleration in the rate of ecosystem conversion

[11] For convenience, the maximum number of plant species is confined to 40,000. For this reason, the data points for Brazil and Colombia lie outside the figure. The two outliers in the upper-middle area represent Australia and the United States.

Figure 2:
Relationship between Biodiversity and Economic Wealth



Note: Economic wealth on the vertical axis is described by GDP per capita (2002) in international PPP$. Biodiversity is described by species richness of higher plants.

Source: Own presentation using data from WRI (2003), World Bank (2004).

is evident. Nevertheless, for specific ecosystems in certain biodiverse regions, the rate of conversion is still increasing (MEA 2000a: 56ff.).

In this context, the change of tropical forest cover attracts particular attention. Tropical forests do not only serve carbon sequestration in climate policy but also represent the habitat of a large part of the living global species richness. Achard

Table 1:
Annual Forest Area Change in Continental Regions, 1990–1997 (in 10^6 hectares per annum)

Continental region	TREES-II study	FAO study
Latin America	–2.2 ± 1.2	–2.7
Africa	–0.7 ± 0.3	–1.2
Southeast Asia	–2.0 ± 0.8	–2.5
Global	–4.9 ± 1.3	–6.4

Source: Achard et al. (2002).

et al. (2002) compare empirical findings on recent tropical deforestation in their own study entitled TREES-II (Tropical Ecosystem Environment Observations by Satellite) with findings by the FAO (2001) (State of the World's Forests). Table 1 summarizes the annual net forest area change in absolute terms for the three continents in the tropical climate zones.

According to these figures, tropical forests shrink at the net rate of 3.6 to 6.4 million hectares per annum. This corresponds to annual deforestation rates of 0.38 to 0.91 percent on a continental scale. For local deforestation hotspots that can be found within each of the three regions, annual rates of 3 to nearly 6 percent are evident (Achard et al. 2002).

Regarding *species* diversity and the difficulties in assessing the total number of living species, it is assumed that many species become extinct without people even noticing since these species have neither been discovered nor described. This is effectively the nature of the problem of determining the extent of extinction. Consequently, there is documented evidence of a comparatively small number of 784 extinctions in recent centuries. It is very likely that this figure only represents a minor portion of the total number of actual extinctions that occurred (Baillie et al. 2004). Furthermore, it is recognized that extinctions have occurred in all periods. Nevertheless, it is estimated that the current rate of global extinctions is 1,000 to 10,000 times higher than the natural background rate of extinction (Barbault and Sastrapradja 1995; MEA 2005b: 3, 21).[12]

Turning to living and comparatively well-described species, several of them have highly decimated populations and are therefore considered in danger of extinction. In its recent assessment, entitled the 2004 IUCN Red List of Threatened Species, the World Conservation Union (IUCN) argues that at least 15,589

12 Background rates of extinction are assumed to be roughly 0.1 to 1.0 extinctions per million species per annum (MEA 2005b: 18). The estimates as to the current rates of extinction are subject to considerable uncertainty (MEA 2005b: 56). The estimates cited refer primarily to extinctions in tropical forests (Barbault and Sastrapradja 1995).

species are currently under the threat of extinction. Regarding the various species groups, 12 percent of birds, 23 percent of mammals, and 32 percent of amphibians are identified as threatened. In addition, indices measuring the trend in the endangerment show deterioration particularly for birds and amphibians. The habitats of endangered species are predominately located in the tropics, especially on mountains and islands (Baillie et al. 2004).[13]

Finally, data on so-called genetic erosion, i.e., loss of *genetic diversity*, is sparse. The importance of genetically diverse organisms for the generation of ecosystem services is not, in fact, assessed in quantitative terms. Anecdotal evidence of decimation in natural diversity is only reported for the area where genetic diversity most visibly contributes to human production, i.e., the use of crop varieties by farmers. Existing figures describe genetic uniformity in selected crops and cultivated areas. Information on the reduction of diversity is obtained, for example, by comparing current inventories of crop varieties with past inventories carried out at the beginning of the 20th century. It turns out that, for many crops, 80 to 90 percent of varieties have irreversibly been lost (Thrupp 2000; FAO 1998: 30ff.).[14]

In light of the described uncertainties, it is sometimes asserted that the *impact of biodiversity loss* on human life may not be as substantial as the public perceives it to be because:

- existing figures, particularly on extinction, are subject to great uncertainty and may potentially overestimate the actual biodiversity loss (e.g., Lomborg 2001), and
- human adaptation cushions much of the impact.

In response to these caveats, it should first be noted that many people care about biodiversity and feel a moral obligation to forgo resource exploitation in favor of preservation. This connection between the perception of biodiversity loss and human well-being is virtually independent of information on the precise extent of extinction but is driven by the evident signs of degradation (OECD 2004: 21).

Furthermore, challenging the accuracy of empirical findings on species loss does imply that less extinction harms human life to a lesser extent. By focusing

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13 While these figures all refer to a specific diversity level, the World Wildlife Fund (WWF) has developed an aggregate measure for the status of biodiversity: the Living Plant Index (LPI) combines data on species extinction and changes in land use cover (WWF International 1999). According to the 2002 assessment, the LPI for terrestrial species has declined more or less progressively to a total extent of 39 percent over the last 30 years.

14 For a critical review of the issue of genetic erosion, see the case study on wheat by Smale (1997).

on the uncertainty regarding the level of species extinction, the role of ecosystem services for human well-being is not addressed. The provision of these services depends upon ecosystem integrity, which in turn is determined by interactions among prevailing species or the survival of keystone species (Kysar 2002; WBGU 2000: 50ff.; Chapin et al. 2000). Therefore, general evidence on species loss does not translate linearly into an adverse impact on human well-being but, rather, serves as an indicator for threatened ecosystem integrity and decreasing ecosystem services.

Regarding the potential strategies to adapt to biodiversity loss and substitute man-made services for degraded ecosystem services, it has to be taken into account that the complexity of natural ecosystems is often poorly understood. Given our incomplete understanding of how ecosystems function, it is difficult to assess whether such strategies are efficient or even viable.[15] The capacity to adapt to and mitigate the adverse impact of biodiversity loss needs to be assessed with regard to specific ecosystem services, preferably against a case study background where the technical possibilities of substitution and costs for adaptation are described more precisely.

In spite of the need for specific information on a case study level, there is general evidence that there is a disparity in the degree to which people are affected by biodiversity degradation. This disparity is driven by the dependence of the different stakeholders on biodiversity for the constituents of their well-being (MEA 2005a: 12f., 2005b: 40f.). Such dependence is described in a comparative analysis of people living in the same region or country, or of people in developed and developing countries.[16] Here, it is often shown that particularly poor countries and poor people are strongly affected because they simply suffer from the lack of substitutions and alternatives to specific ecosystem services, as well as the lack of (access to) supporting institutions that could compensate for a drop in well-being resulting from biodiversity degradation. To offset this loss in well-being, poor people often use natural resources beyond sustainable limits, leading to the further loss of biodiversity (WBGU 2005: 82ff.). Finally, this biodiversity-poverty nexus raises skepticism about whether there are widely available possibilities to adapt to biodiversity loss.

To resume, the asserted caveats on the importance of biodiversity loss are not sufficient to downgrade the current priority of biodiversity conservation on the

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15 Natural scientists point out that, under certain circumstances, ecosystem services are vital for an ecosystem's integrity and therefore cannot be replaced by man-made services without risking drastic negative repercussions for the ecological system and human life in particular.

16 A further investigation of disparities concerns the comparison of the adaptive capacity of future generations compared to that of the present generation. Such an investigation, however, relies on projections about the future.

international policy agenda. On the contrary, the evidence of the continuing and even accelerated loss of biodiversity supports the notion that people should make more of an effort in this regard.

Causes of Biodiversity Loss

International policies to slow down the loss of global biodiversity and support its preservation have to concentrate on the causes of its degradation. Many studies have addressed the identification of theses causes (e.g., Armsworth et al. 2004; Chapin et al. 2000; Sala et al. 2000). The following five phenomena are typically mentioned as the *direct drivers* of biodiversity loss. They all relate to *anthropogenic interference* in the natural environment (MEA 2005a: 14ff.):

Land use change as a driving factor means loss of habitat, which is attributed to (1) the conversion of natural, previously unmanaged areas to cultivated land, (2) the degradation of the habitat quality by human use, and (3) fragmentation, i.e., due to the conversion of single parts, the habitat becomes increasingly patchy, with the consequence that the conditions for species' living and survival are affected (Armsworth et al. 2004).

Climate change, resulting from anthropogenic greenhouse gas (GHG) emissions, is supposed to increasingly affect biodiversity. Changes in the atmospheric GHG concentration are associated with the projected increases in mean temperatures, changes in patterns of precipitation, and a rise in the sea level. These are supposedly changing present ecological processes in many ecosystems. While the population and biomass production of specific species in certain regions will benefit from this climatic development, recent evidence and scientific projections imply that climate change will strongly affect the overall species richness: up to one third of the species living on earth may not be able to migrate to another habitat or adapt to the change in climatic conditions, with the consequence that the risk of their extinction will drastically increase (Thomas et al. 2004; Root et al. 2005; WBGU 2000: 253ff.).

Invasive species are exotic species that invade a new environment where they manage to persist and propagate. As a result, the prevailing community of native species is often affected in that their population shrinks and, consequently, some of these species become extinct. Humans have often introduced invasive species deliberately without considering or predicting the adverse ecological impact of doing so. Otherwise, invasions have frequently happened accidentally as a by-product of the increasing trading activities in the globalized world (Sala et al. 2000; WBGU 2000: 194ff.).

Overexploitation, or, synonymously, over-harvesting refers, to the intensive extractive human use of specific species. When the population of such species declines, other species that depend upon the exploited species as an element of the biological food chain may also be affected. As a result, the functioning of the ecosystem, as well as the provision of ecosystem services, can be disturbed (Armsworth et al. 2004).

Pollution refers to the human deposition of nitrogen and phosphorus (eutrophication) in ecosystems, leading to various changes in the living conditions of different organisms. These changes occur in the form of a gradual increase in nutrients in ecosystems. Such nutrient enrichment aims at promoting the biomass production of certain (commercial) plant species at the expense of other species. The induced change in an ecosystem's species composition can affect the provision of the ecosystem services, such as water purification or climate regulation (Sala et al. 2000; MEA 2005a: 15f.).

Focusing on a specific ecosystem, for example, from a case study perspective, the individual direct factors may not be relevant to the same extent. Moreover, the impact may vary according to the form of ecosystem (biome) and the specific socioeconomic environment in which the ecosystem is embedded. On a biome level, the intensity of the impact and trend for the main drivers of change is differentiated according to the different forms of biomes (MEA 2005a: 52). It is evident that habitat change represents the dominant driver in most biomes and that its importance is supposed to continue or even increase in the future. Furthermore, climate change currently bears a low to moderate impact on biodiversity in nearly all biomes. However, recent studies project that the significance of climate-induced changes in biodiversity will increase rapidly in the future.

Typically, these direct drivers have to be considered against the background of *indirect drivers* of ecosystem change and biodiversity loss, i.e., factors that influence the human decision on resource uses. Biodiversity policies have to take into account these underlying factors in order to be effective. Although there is no unique classification, five indirect drivers are often named in the literature: (1) demographic development, (2) economic factors, (3) technological change, (4) policy and institutions, and (5) cultural environment.

Typically, these drivers act synergistically (Geist et al. 2002; MEA 2005a: 64ff., 2005b: 58ff.).The qualitative importance and empirical significance of the individual factors is an ongoing subject of research. In principle, ecosystem change can occur in different forms and is, depending upon the form of change, difficult to measure in quantitative terms. A major focus in empirical research has been deforestation, which is a dominant form of change in the developing countries and which is comparatively well described (Angelsen and Kaimowitz 1999; Barbier and Burgess 2001; van Kooten and Bulte 2000: 441ff.). Accordingly, deforestation forms the background to the subsequent discussion on the indirect drivers of ecosystem change:

Demographic development refers to the fact that, as mentioned earlier, developing countries in particular are experiencing substantial growth in population, leading to the increased consumption of ecosystem services. It is believed that this rapid population growth initiates changes in land use and ecosystem degradation. Aside from population growth, it has been observed that migration frequently happens at a faster rate. Migration affects biodiverse ecosystems when occurring as colonization and agricultural expansion in previously unmanaged natural areas (Marcoux 2000). In the literature, it is argued that local or regional population growth may not be an exogenous factor in habitat change but, rather, interdependent on the natural conditions and human use in the considered ecosystem (Angelsen and Kaimowitz 1999).[17]

Various *economic factors* are considered indirect drivers of ecosystem change. Parameters that influence the individual decisions on ecosystem management and use of resources can be distinguished from parameters that are determined by broader, economy-wide forces. The former category is described by prices of agricultural/forestry input and output (including taxes and subsidies), credit available to the landowner, and wages in alternative employment. The latter category encompasses absolute income levels (poverty) and macroeconomic parameters, such as the available capital stock (infrastructure), economic growth, and foreign trade (including foreign debt and the degree of trade liberalization).

For these parameters, the net impact on ecosystem change is investigated both on an analytical and empirical basis. Analytical studies attempt to identify the direction of the impact caused by the different parameters; however, a clear direction is only found for some parameters. Empirical studies suggest that the direction and significance of the impact differs among parameters. Regional particularities seemingly have a substantial influence (Angelsen and Kaimowitz 1999; Barbier and Burgess 2001; van Kooten and Bulte 2000: 441ff.).[18] In general, interaction between the different parameters often reinforces the total net impact on the man-made changes to ecosystem but sometimes interaction can have an offsetting influence.

Technological change, which could well be subsumed under economic factors, refers to the development and diffusion of scientific knowledge of the functioning of an ecosystem, as well as the practices of its management. This knowledge

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17 Generally, population growth influences the endowment of the labor force and therefore has an impact on factor prices and the level of household income. In this regard, the population issue is sometimes subsumed under economic factors.

18 In spite of the uncertainties and ambiguities, both analytical and empirical studies imply that by increasing output prices in agriculture and forestry, the conversion of biodiversity-abundant habitats in developing countries generally increases, while increasing wages in alternative sectors generally has an offsetting influence (Angelsen and Kaimowitz 1999).

is present in technologies that predominately serve the management and/or exploitation of ecosystem services. Technological change, i.e., the development of existing technologies on the one hand leads to more rapid changes to the ecosystem as new technologies reduce the costs of resource depletion or support and enhance only selective ecosystem services (intensification); on the other hand, technological change favors the preservation of biodiversity by replacing poor technologies for resource use with technologies that support sustainability (Geist et al. 2002; MEA 2005a: 66). In addition, new technologies that have no direct connection to the management of ecosystems indirectly assist conservation. This is because new technologies lead to an increase in the economy's aggregate supply that, in turn, reduces output prices and thereby contributes to a reduction in the pressures for ecosystem change (Angelsen and Kaimowitz 1999).

Policy and institutional factors are related to the humanly devised framework of ecosystem management and encompass formal rules and governance structures. Conditions like public participation in decision making, mechanisms of conflict resolution, education, or the role of women in civil society are supposed to have an impact upon ecosystem change (MEA 2005a: 64f.). While the significance of some of these parameters is frequently investigated in other social sciences, economic studies mainly focus on ecosystem change as a consequence of the prevailing property rights regime, particularly in the context of land tenure arrangements, as well as of price distortions resulting from policy failures. On a broader scope, the impact of policies on structural adjustment and trade liberalization is studied. Although the analysis of these parameters is frequently constrained by the data availability of institutional indices, results from empirical studies imply that institutional factors have an influence on ecosystem change (e.g., Barbier and Burgess 1991).

Finally, *cultural aspects*, i.e., attitudes and values, as well as religious beliefs, are assumed to have an impact, since they influence human behavior towards resource consumption and the commitment to preserving the environment. In this respect, missing values and public unconcern can favor ecosystem change and biodiversity degradation. (Geist et al. 2002; MEA 2005a: 65). The findings in empirical studies support this hypothesis.

2.2 International Policies for the Protection and Conservation of Global Biodiversity

The policies to slow down and stop the loss of biodiversity can be justified on *ethical grounds* by pointing to the environment's intrinsic values and the rights of future generations. However, from the perspective of the present generation, such policies can also be desirable in order to prevent or contain *welfare losses* due to the degradation of biodiversity and a reduction in ecosystem services (Perrings and Opschoor 1994; WBGU 2000: 333).

To attain effective conservation, policies ideally address the threats to biodiversity, i.e., the described direct and indirect drivers of ecosystem change. Since the individual drivers of change are pronounced to varying degrees in ecosystems, appropriate policies are formulated according to site-specific threats. In formulating such policies, biodiversity typically represents a crosscutting issue that biodiversity conservation is closely linked to several policy areas aside from nature protection in a narrow definition. More precisely, biodiversity policies are often subject to a complex and sometimes contradictory framework of several domestic and international regulations aimed at different objectives, for example, in the areas of agriculture, tourism, residential development, or transportation infrastructure. Therefore, effective policies to control ecosystem change and conserve biodiversity include the coordination of the multiple instruments that regulate or control economic activities in the biodiverse ecosystems and their socioeconomic environment.

The decisions on these policies, their design and implementation, have to be made against the background of significant *uncertainties* (or lack of knowledge) regarding functional relationships between biodiversity components, as well as the interaction between the ecological and economic system. More specifically, uncertainties result from gaps in information concerning, inter alia, supporting ecosystem services, as well as cultural and regulating services, particularly in as far as they are not evaluated in a market framework (MEA 2005a: 101f.).

Policy formulation and subsequent implementation is also influenced by the fact that actors benefiting from biodiversity policies may not coincide with the actors carrying the costs of conservation. This is mainly due to nonexclusive provision of various ecosystem services, which are sometimes also nonrival in their use. For *national policies*, the resulting conflict of interests can be resolved within domestic institutions. However, for many ecosystem services, benefits flow across borders and therefore decisions regarding their maintenance should be addressed in *international policies*. In contrast to the domestic level, international policies are governed by the autonomous decisions of the sovereign countries involved. In order to reconcile the countries' diverting interests on conservation and extractive ecosystem uses, cooperative solutions have to be found.

On a global level, the diverting interests among countries are reinforced by the described asymmetry between economic wealth and endowment with biodiversity. From a simplified perspective, there are developing countries that host major parts of global biodiversity and that have a strong interest in fostering national economic development by exploiting their own natural resources on the one hand—even though this can harm certain ecosystem services; on the other hand, developed countries with few biodiversity endowments of their own often depend upon both tradable and nontradable ecosystem services that are provided in the developing world and that may be endangered by degradation.

Against this background, cooperation between countries is taking place in several multilateral fora and has resulted in the negotiation and ratification of several *international environmental agreements* (IEA) that establish the framework for the design and implementation of cross-border or global biodiversity policies. Table 2 names previous international agreements that address the conservation of wildlife and nature in a narrow definition. The agreements appear in descending order according to the number of signatories. The first five agreements in the table, as well as the last one, show a global scope, while the remaining ones are restricted to conservation on a continental or regional level. In addition to these agreements (and the many others that show a comparatively smaller number of signatories and that are not mentioned here), the protection of biodiversity is also addressed in international agreements concerning other areas of environmental protection and sustainable resource uses. This includes, for example, agreements on forestry, plant protection, fisheries, international rivers, air pollution, or the transportation of hazardous materials (Barrett 2003: 165ff.).

Regarding ecosystem change on a global level, four conventions are of particular importance. Aside from the CBD, the Convention on Wetlands, and the Convention on Migratory Species, there is the United Nations Convention to Combat Desertification in Countries Experiencing Serious Drought and/or Desertification (MEA 2005a: v).[19]

The old and more recent international agreements together build a grown and developed regime for international and national biodiversity policies. Previously, the single agreements have typically pursued an approach that more or less addresses *individual threats to biodiversity*, for example, overexploitation induced by increasing international trade, without taking into account the interplay between the many drivers of ecosystem change and biodiversity loss. Several agreements are also *regionally focused*, i.e., they only address ecosystems in Europe or Africa but not the interactions between ecosystems on an intercontinental level. Finally, previous agreements concentrated on rather narrowly defined elements of biological diversity, such as *species* (particularly animal and plant species) or certain *forms of ecosystems* (e.g., wetlands).

The 1992 CBD differs considerably from its preceding agreements in that it pursues an approach that is *comprehensive* and *integrated* with respect to both ecological and socioeconomic features of biodiversity conservation. In this re-

19 The Convention to Combat Desertification was signed in 1994. Presently, 179 countries are parties to the convention.

Table 2:
Selected International Environmental Agreements on Nature and Wildlife

Agreement (with global or regional scope)	Date of adoption	Number of signatories
Convention on Biological Diversity	1992	178
Convention Concerning the Protection of the World Cultural and Natural Heritage[a]	1972	160
Convention on International Trade in Endangered Species of Wild Fauna and Flora	1973	155
Convention on Wetlands of International Importance	1971	124
Convention on the Conservation of Migratory Species of Wild Animals	1979	75
The Antarctic Treaty[a]/Protocol to the Antarctic Treaty on Environmental Protection[a]	1959/1991	44/38
Convention on the Conservation of European Wildlife and Natural Habitats[a]	1979	44
African Convention on the Conservation of Nature and Natural Resources[a]	1968	43
Convention of Nature Protection and Wildlife Preservation in the Western Hemisphere[a]	1940	22
Agreement for the Conservation of Bats in Europe[a]	1991	21
International Convention for the Protection of Birds	1950	15

Note: Agreements are ordered by number of signatories. Numbers of signatories according to Barrett (2003).
[a]Membership is restricted.

Source: Barrett (2003: 165ff.).

spect, the CBD has become the major framework for most international and national biodiversity policies and supports their overall coherence (Swanson 1997: 79ff., 1999).

It is commonly understood that the provisions of the CBD are guided by the principle of *sustainable development*, meaning the compatibility and integration of environmental protection and economic development. In this vein, incentive measures are to be introduced that induce and support self-interest in efforts to conserve globally important biodiversity. The proper definition and assignment of property rights to the resources connected to biodiversity is considered crucial in this context (WBGU 2000: 262f., 337f.).

The *objectives* supported by these principles are listed in Art. 1 of the CBD. The triad of objectives encompasses:

- the "conservation of biological diversity,
- the sustainable use of its components, and
- the fair and equitable sharing of the benefits arising out of the utilization of genetic resources."

In Art. 6 to 14 of the CBD, the *measures* for attaining these objectives are broadly described and formulated. They include, inter alia, the development of national strategies for conservation and sustainable use, the establishment of a system of protected areas, activities to monitor biodiversity components and identify adverse impacts, and the adoption of incentive measures influencing and controlling human resource use.

The CBD regime is manifested in several *institutions*. First, this is the Conference of the Parties (COP) (Art. 23 CBD), which is to be held at regular intervals. The COP has the task of keeping under review the implementation of the convention and, if necessary, setting out policy, strategy and program priorities. Furthermore, the COP can adopt amendments or protocols to the convention (Art. 28 CBD) that arrange for the more detailed regulation of specific aspects related to the CBD's objectives. In this respect, the Cartagena Protocol on Biosafety that addresses the safe transfer, handling, and use of living modified organisms resulting from modern biotechnology and that may cause adverse effects on biodiversity was signed in 2000.

Further institutions of the CBD regime exist but are not described in detail here: the Secretariat of the Convention ensures the functioning of the multilateral cooperation (Art. 24 CBD). The Subsidiary Body of Technical and Technological Advice (SBTTA) serves the COP by reporting or providing specific assessments (Art. 25 CBD). The Clearing House Mechanism (CHM) coordinates the supply and demand of biodiversity-related information and thereby facilitates international technical and scientific cooperation (Art. 18(3) CBD). The Global Environment Facility (GEF), which serves as the CBD's financial mechanism, is studied in more detail in Chapter 4.

Regarding the de jure *acceptance* of the CBD, the agreement has been signed by nearly the entire community of states. Only very few countries have not yet ratified and translated the provisions into national law. Nevertheless, one of these is the United States, which objected to the CBD clauses regarding intellectual property protection in biotechnology. The de facto acceptance, i.e., the proper implementation of the CBD's provision can best be described in qualitative terms.[20] The reasons for potential shortcomings in the implementation of the agreed-upon provisions are a lack of political will on the part of the country

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20 Concerning implementation, it has been pointed out that, although the CBD assigns property rights on an international level, problems in the allocation of property rights and enforcement on a national level are not and cannot be pursued in this framework. Since it is recognized that for effective conservation, the interests of the individuals who are most concerned with conservation measures need to be taken into account, this demands particularly the participation of indigenous people and local communities in the political decision-making process (Swanson 1997: 79ff.).

hosting valuable biodiversity endowments and/or a lack of national governance and institutional capacities.[21]

Aside from practical problems in the implementation, it can be questioned as to whether the CBD correctly addresses the key issues in the context of biodiversity loss and proposes the right priorities and appropriate measures for conservation accordingly. The answers depend upon how the progress of previous and existing biodiversity policies is assessed. Recent developments in international fora provide an implicit assessment in this regard.

In 2002, i.e., ten years after the signing of the CBD, the sixth COP adopted the "Strategic Plan for the Convention on Biological Diversity", which commits the countries to a more effective implementation of the CBD's objectives. More specifically, it aims "to achieve by 2010 a significant reduction of the current rate of biodiversity loss on a global, regional and national level". This "*2010 target*" is specified in more detail in several focal areas where for each of them suitable indicators are to be developed in order to describe and assess the impact of biodiversity policies (MEA 2005b: 77ff.).

In the same year, the "2010 target" was endorsed by the United Nations' World Summit on Sustainable Development (WSSD) in Johannesburg. In addition, various international fora acknowledged the important relationship between biodiversity and the *Millennium Development Goals* (MDG), established in 2000 by the United Nations' "Millennium Project" (UNDP 2005; WBGU 2005; MEA 2005a: 61f.).

Moreover, given the assumed impact of poverty on the use of natural resources and on biodiversity loss, it is also recognized that biodiversity conservation and sustainable use are important elements in achieving the MDG. Important contributions of biodiversity are particularly evident in

- eradicating extreme poverty and hunger (e.g., food security).
- addressing several health issues, such as the improvement of maternal health, the reduction of child mortality, and the combat of infectious diseases (UNDP 2005).

Regarding the implied need for jointly addressing biodiversity conservation and the alleviation of poverty, certain policies are considered to stand at the interface of the poverty and biodiversity issues. These include policies aimed at (1) the fair and equitable sharing of the benefits arising from genetic resources, (2) the development of sustainable ecotourism, and (3) international transfers for providing improved/alternative livelihoods in/outside vulnerable/protected eco-

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21 Regarding the latter impediment, there have recently been multilateral efforts to assist developing countries in particular in implementing the necessary capacities for an effective national biodiversity policy.

systems (WBGU 2005: 132ff.). I analyze some of these aspects in the remainder of this study.

Returning to the CBD, it should be mentioned that the alleviation of poverty is not an (explicit) objective of the convention. Although the nexus between biodiversity and poverty is recognized, no concrete actions towards the alleviation of poverty are provided for (WBGU 2005: 132f.). The current challenges for biodiversity policies are evident in overcoming obstacles to the CBD's implementation in various areas (Strategic Plan on the CBD). The following analysis concentrates on selected economic obstacles that occur on an international level.

2.3 The Economics of Biodiversity

Biodiversity policies rely on extensive and profound information regarding the (dynamic) structure and functioning of the ecological system and interplay between the ecological and the socioeconomic system. In this respect, several disciplines are entangled in the investigation of biodiversity. The natural sciences have traditionally been the leading and dominant disciplines.

Because of the multidimensional and multifaceted character of biodiversity, there is an increasing awareness that research on biodiversity needs to pursue an interdisciplinary and integrated approach (Jürgens 2001). The growing literature on the economics of biodiversity conservation and sustainable use adds a socioeconomic element to this research. In general, the strengths of the economic approach can be seen in modeling human behavior in ecosystem use (1) to identify the forces of ecosystem change, (2) to assess the impact of change on human well-being, and (3) to design policy instruments aimed at controlling ecosystem change.

In this context, economic analysis is highly dependent upon the knowledge of ecological science, in particular of relationships describing the interactions between the various ecosystem constituents and the provision of ecosystem services (Armsworth et al. 2004; Chapin et al. 2000; Mäler 2000). In the following sections, I provide a brief overview of the questions asked and approaches used by economic biodiversity research.

2.3.1 Biodiversity and Ecosystem Services from an Economic Perspective

Methodological tools from both *environmental* and *resource economics* are applied to study biodiversity. Nevertheless, there is no unique economic approach to conceptualize biodiversity. Existing approaches have developed around the concept of *ecosystem services*, which establishes the link between biological diversity on the one hand and human well-being on the other hand (see Section 2.1.1).

The fact that the provision of ecosystem services supports human well-being but that human interference in ecosystems simultaneously leads to the degradation of these services (see Section 2.1.2) indicates the prevalence of relative *scarcity*. The demand for resources needed for the generation of ecosystem services exceeds the actual availability. Against the background of alternative options for the management of ecosystems and the natural resources hosted by ecosystems, it is scarcity that motivates the application of methods of economic analysis.

From an economic perspective, ecosystem services are typically perceived as a flow of benefits (Costanza et al. 1997; Heal and Small 2002). The ecosystem is regarded as the asset or stock that generates this flow (Mäler 2000). The stock is conceptualized as *land* or alternatively, as *(natural) capital*.

The land-based approach typically equates a particular ecosystem with land as input in production and pursues a comparison of the net returns from alternative ecosystem uses (*land uses*). The subject of study is the (intertemporal) decision on the allocation of land between competing uses (e.g., Barbier and Burgess 1997; Endres and Radke 1999; Hartwick 1992; Parks et al. 1998).

In a conventional analysis, land is considered a contiguous area representing the micro level (household, village) or macro level (regional/national economy) of decisions on land use. Spatial aspects of land use sometimes enter the analysis by expressing costs and benefits in terms of the distance to a specific location (Cervigni 2001: 39ff.; van der Veen and Otter 2001).

More recent (applied) studies take into account heterogeneity in ecological and economic properties among many adjacent parcels of land. Complex analytical tools and simulation methods are used to describe the spatial structure of land use resulting from either centralized or decentralized decision making (Costello and Polasky 2004; Polasky et al. 2001; Irwin and Bockstael 2002).[22]

According to the natural capital approach to biodiversity, an ecosystem is considered a class of (natural) capital asset that supports economic production and human well-being. The stock of natural capital is subject to deliberate or accidental human interference that either increases or diminishes the size of the stock (Heal and Small 2002; Mäler 2000). In this way, principles of economic theory on *capital* and *finance* are introduced into the analysis of the environment. It is argued that this procedure also shifts the focus from maintaining the existing natural resource base to its active management in order to generate and sustain

22 Van der Veen and Otter (2001) provide a survey on the conceptualizing of land uses and the connections to regional economic theory.

a (natural) capital return. Resource management in this respect may include the accumulation (investment) and liquidation (dissaving) of the resource base (Ekins et al. 2003; Heal and Small 2002; Arrow et al. 1995).

When applying instruments of standard capital theory to ecosystem management, the specific nature of ecosystems relative to other classes of capital assets needs to be taken into account. This means sensitivity in the provision of ecosystem services to abiotic parameters, such as the landscape structure, and complex biotic interaction within ecosystems—including human interference that leads to unpredictable repercussions (Heal and Small 2002). Overall, the strengths of the concept of natural capital are the support of the integrated investigation of the ecological and socioeconomic dimensions of managing ecosystems (Chiesura and de Groot 2003).

The two conceptual approaches are not mutually exclusive. Moreover, in the economic literature on biodiversity and ecosystem management, elements of the two concepts are combined. This is indicated, for example, by the role of land as a "base resource" for biodiversity components (Swanson 1994) or the recognition of man-made input in the provision of ecosystem services as forms of investment (Blandford and Boisvert 2002).

Integration of Ecological and Economic Analysis

Economic analysis conceptually describes the linkage between human activities and the environment by various input output relationships. The environment, or, synonymously, the ecological system, represents the background of human activities. The system itself is quite large and does not operate at its physical boundaries. Although this system may, in principle, exist in alternative states, it evolves slowly compared to the dynamic economic system. In this perception of a stable natural environment, analytical economic studies typically regard the economic system separately from the ecological system (Limburg et al. 2002).

In the literature, it is sometimes argued that under various circumstances, this perception does not seem appropriate: instead of stability in the environmental conditions, the ecosystem's composition and, thereby, the provision of ecosystem services changes simultaneously with economic actions. In analytic terms, the economic and the ecological system are jointly determined. This fact has put forward calls for the *integration* of ecological and economic (production) principles in the analysis of ecosystem management (Batabyal 2000; Young 1992: 35ff.).

Studies in this regard typically consider a *dynamic bioeconomic framework* in order to describe an ecosystem in a stylized way (Siebert 1983: 110ff.).[23]

[23] The building blocks therein are differential equations with the state variables that are interpreted as the resource stock or, alternatively, as environmental quality represent-

Traditionally, the focus falls upon the production of a homogenous biomass or the stock/population of a single species. In addition to this monoculture concept, the dynamics of an ecosystem can be represented by way of interaction between multiple species. Regarding a closed community of species located in an ecosystem, a set of differential equations can be used to describe fluctuations in individual populations that evolve endogenously and depend upon the size of the other species' populations. These dynamics lead to changes in species composition, including the eventual fate of some species (Armsworth et al. 2004). The composition of species in turn influences the flow of ecosystem services provided.

While these changes occur even in the absence of direct anthropogenic interference, human resource management deliberately or accidentally influences the processes within an ecosystem.[24] With the analytical modeling of these processes, alternative states of an ecosystem are described.

Regarding the functional relationships representing the ecosystem, ecological empirical studies reveal that the dynamics in many cases correspond to transformation possibilities that constitute a nonconvex set. This *nonconvexity* is often due to feedback processes that are, in turn, induced by discontinuities represented by *ecological thresholds*: within a range of human-induced or natural perturbations in an ecosystem, the flow of generated services changes in smooth reactions. When perturbations gradually lead to the crossing of the threshold value of specific ecological variables, the size of the flow changes at once in a discrete, discontinuous way (bifurcation) (Dasgupta and Mäler 2003; Mooney et al. 1995). The magnitude of this change and the location of the threshold are frequently uncertain or unpredictable (Levin et al. 1998; Limburg et al. 2002).[25]

Furthermore, crossing an ecological threshold is often *irreversible*: once perturbations have caused a decrease/an increase in the flow of ecosystem services, a step-wise reversal of the induced ecosystem changes may again lead to smooth, continuous reactions in the size of these services. However, the overall level of the services is below/above that of the original before the

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ing specific ecosystem services. The ecosystem dynamics are analytically presented by a regeneration function (Siebert 1983: 110ff.).

24 In order to analytically describe ecosystem dynamics in a tractable way, the models in the literature are frequently defined deterministically (Dasgupta and Mäler 2003). Alternatively, for circumstances characterized by a high degree of uncertainty or complex, poorly understood ecological phenomena, a stochastic modeling approach is considered more suitable (Perrings 1998).

25 Empirical examples for perturbations leading to nongradual ecosystem changes are overgrazing, leading to desertification, overexploitation in marine ecosystem, leading to the collapse of fishery, or habitat fragmentation, leading to species extinction (Levin et al. 1998).

perturbations occurred (hysteresis) (Dasgupta and Mäler 2003; Mooney et al. 1995).[26]

With regard to the dynamics in an ecosystem moving between alternative states, it is questioned as to which states are actually persistent and what flow of ecosystem services is connected with the stable states of system—provided that at least one exists. In ecology, several concepts exist in order to describe *stability* (McCann 2000). In integrated economic-ecological research, stability is typically characterized by the concept of *resilience.*[27] Resilience refers to the capacity of an ecosystem to (1) recover from perturbations and return to its initial state or (2) to absorb perturbations before switching to another state (Perrings 1998).

In order to characterize resilience, it is recognized that ecosystem dynamics take place in a *hierarchical structure*: each system is described by a structure of subsystems nested within each other.[28] Each subsystem shows its own speed of adjustments at different spatial and temporal scales and hence is subject to its own stability properties in the sense of resilience/adaptive capacity. Regarding the processes at the different levels, the *connectedness* of subsystems over space and time is highlighted, meaning that localized short-term perturbations, for example, can already contribute to long-term changes in large-scale systems (Levin et al. 1998; Perrings 1998; Perrings and Walker 2004).[29]

The determinants of resilience/ecosystem stability are not easy to generalize and are subject to research. In short, empirical evidence indicates that—aside from other factors—there is a certain connection between ecosystem *stability and diversity* in living organisms. In an alternative concept, empirical findings imply that stability depends to some extent upon the prevalence of a set of specific biological species, i.e., *keystone species* (Chapin et al. 2000; Folke et al. 2002; McCann 2000).

The connectedness of subsystems highlights the impact of spatial and temporal variations on ecosystem dynamics: temporal variations refer, inter alia, to the impact of abiotic factors describing *climatic/seasonal conditions*. Spatial variations are differences in the *spatial structure* of a landscape, including topography, hydrology, or vegetation cover. These parameters have an influence upon the population sizes of species and, consequently, the species composition within an ecosystem, which in turn determines the provision of ecosystem services (Armsworth et al. 2004). In addition, attention is drawn to the spatial phenomena

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26 Irreversibility also occurs in the absence of thresholds (Dasgupta and Mäler 2003).

27 An alternative but related concept of stability is the concept of persistence (Batabyal 2000).

28 For example, in forest ecosystems, the hierarchy includes the level of leaves/needles, the stand, the forest, and, finally, the biome (Perrings 1998).

29 For an analytical conceptualization of these interactions, see, for example, Perrings (1998), Perrings and Walker (2004), or Batabyal (2002).

of the *migration* of species between disconnected or fragmented habitats (metapopulation theory) (Sanchirico and Wilen 1999).

Finally, temporal and spatial variations are assumed to interact in complex ways when exerting an influence upon ecosystem dynamics and ecosystem services. Examples of the analytical modeling of spatial and/or temporal variations on populations and the community composition within ecosystems are given in Brock and Xepapadeas (2002), Sanchirico and Wilen (1999), and Tilman et al. (2005).

Generally, the roughly described properties or "building blocks" of ecosystem dynamics and resilience translate into constraints that need to be taken into account when an ecosystem is managed for economic objectives. The conceptualization for economic analytical purposes is generally located in an area of tension between an appropriate representation of the ecosystem functioning and a necessary reduction in complexity. A more complex conceptualization is required when nonlinearity noticeably affects management options. Nevertheless, human interference (perturbations) within a considered range often only leads to marginal changes in the flow of ecosystem services ("marginal regime")(Limburg et al. 2002). In such cases, comparatively simple modeling may serve as an appropriate approximation.

2.3.2 Valuation of Ecosystem Services

Given that there are alternatives for ecosystem/resource management, rational decision making requires well-informed decision makers. Since resource management refers to human interference that either aims to support or prevent undesirable changes in ecosystem conditions, information on the net benefits derived from the flow of services connected with each of the alternative states of the ecosystem is required.

In general, *market prices* for goods and services from ecosystems represent a good source of information. In two respects, however, prices only provide incomplete information on the *value* of ecosystem services, i.e., their contribution to human well-being:

First, existing prices often understate the true value because of policy failures that support price distortions, which are detrimental to sustainable resource management. Market prices in this regard do not indicate the true scarcity of these services. Furthermore, prices only reflect the preferences of present generations that are able to participate in market interactions. The benefits derived by marginalized people or future generations are not represented. It is also of note that prices are subject to the existing distribution of monetary and nonmonetary endowments (Heal 2000; Pagiola 2004: 15).

Second, market prices do not exist for those ecosystem services that represent pure or impure public goods and that are therefore not traded and valued in a market framework. Nonmarket values are an important factor in resource management. By focusing upon market values only, the resulting incomplete assessment of the values of ecosystem services favors decisions where non-sustainable resource use leads to biodiversity loss (Krutilla 1967).

The Total Economic Value of Biodiversity

Given the information gap with respect to values, economic theory provides a set of methods to measure the (nonmarket) value of a flow of ecosystem services in monetary terms. The fact that multiple ecosystem services are provided simultaneously motivates an assessment of biodiversity that includes all categories of values associated with its conservation and/or use. For this reason, the concept of *total economic value* (TEV) has been put forward (Moran and Pearce 1997).

Due to its welfare economic foundations, the concept of TEV puts an anthropogenic perspective on ecosystem services, meaning the instrumental values of biodiversity drive its human use, including measures for its preservation (Fromm 2000; Hampicke 1999). Figure 3 recapitulates the categories of TEV.[30] This concept is used to estimate either the total benefits derived from an existing flow of ecosystem services or the change in net benefits resulting from a change in that flow. In evaluating specific resource management, the latter approach provides the relevant information (Pagiola et al. 2004: 15ff.).

Aside from the TEV concept, various perspectives on the value of biodiversity can generally be taken (Nunes and van den Bergh 2001). In contrast to the categorization represented in the figure above, values can be aggregated and classified according to classic welfare economics as *production values* and *individual/consumpton values*. A third category of value is then generated by the ecosystem's ecological integrity. Drawing upon the categories of ecosystem services described in Section 2.1.1, the values of ecological integrity refer to the supporting ecosystem services, which assist provisioning, regulating, and cultural services that in turn generate the direct use and nonuse values (MEA 2005b: 19). In this regard, the values of ecological integrity are referred to as *primary values*. They are distinctive from the derived secondary values, which largely coincide with the instrumental values described in the TEV framework (Gren et al. 1994; Fromm 2000).[31]

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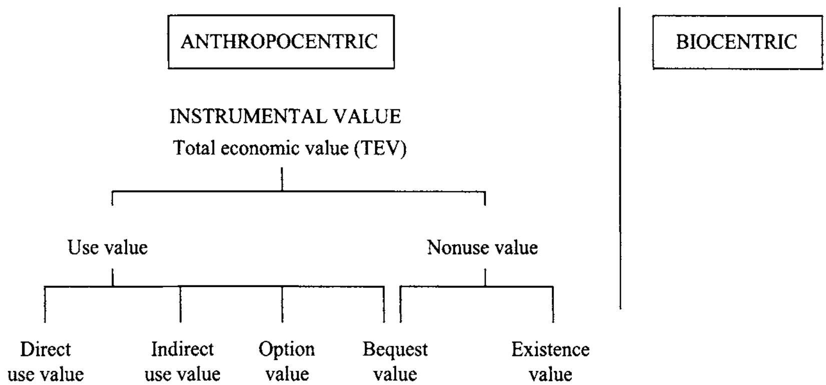
30 Note that the bequest value of biodiversity is categorized differently across the studies.

31 Further names assigned to this type of value are "indirect use value," "inherent value," "infrastructure value," and "contributory value" (Fromm 2000; Nunes et al. 2001). Regarding the "indirect use value" in the TEV framework, Fromm (2000)

Figure 3:
A Typology of Biodiversity Values



Source: Pagiola et al. (2004: 9), Bateman et al. (2003: 2), Barbier (1997: 135).

In addition to instrumental values, it is recognized that people assign values to biodiversity in its own right (Bateman et al. 2003). This view is based upon biocentric considerations. Generally, the value that resides in nature has been the subject of an extensive conceptual debate in the disciplines involved. For reasons of space, this debate is not recapitulated here (Faber et al. 2002; Gowdy 1997; Turner et al. 2003).

Applied Economic Valuation of Nonmarket Ecosystem Services

Returning to the instrumental values of ecosystem services, various *valuation methods* can be applied to ascertain the value that people assign to these services. In general, two classes of methods can be distinguished. The first class is represented by the *revealed preference* valuation method. It includes the travel cost method, the hedonic price method, the replacement cost assessment, avert behavior, and the production function approach. These methods all have in common the fact that they rely on existing market data on goods and services related to biodiversity. The second class is represented by the contingent valuation method. In contrast to the former class, numerical values are determined on the basis of *stated preferences* (de Groot et al. 2002; Nunes and van den Bergh 2001).[32]

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argues that it differs from the concept of primary values, since ecological structures are not explicitly accounted for.

32 Although alternative descriptions of the methodological toolbox exist in the literature, the classification described includes its major elements. For minor amend-

The available techniques can be assigned to the categories of biodiversity values. The application of the methods of revealed preferences is often limited to specific classes of values. For example, the travel cost method is typically confined to valuing cultural ecosystem services. In contrast, the contingent valuation method based upon stated preferences can easily be applied to nearly all classes of values. It only fails to describe the primary values of biodiversity, since it is assumed that people are typically unaware of these values or rarely experience them directly (de Groot et al. 2002; Gowdy 1997; Nunes and van den Bergh 2001).

Overall, numerical measures of the values of ecosystem services derived from applied works are generally not considered univocal and unambiguous. Moreover, their dependence upon the level of diversity, the form of value, its socio-economic context, as well as the applied valuation method, is recognized (Nunes and van den Bergh 2001; Turner et al. 2003).

Regarding applied valuation within the framework of the TEV of biodiversity, there is a vast number of *case studies* on valuing the flow of ecosystem services or the expected changes in this flow. However, most of the studies do not provide a complete assessment of all services along the lines of TEV (Pagiola et al. 2004). The focus of the studies typically falls upon valuing single or multiple ecosystem services generated in a specific but local context (Barbier and Strand 1998; Carson et al. 1994). Frequently, the valuation in this respect represents the integral part of a cost benefit analysis, meaning the derived values are compared with the costs of conserving/restoring the considered ecosystem. An overview of these empirical studies is given, for example, in Cartier and Ruitenbeek (1999), Nunes and van den Bergh (2001), Pigiola et al. (2004), and SCBD (2001).[33]

In contrast to this valuation on a *micro level*, Costanza et al. (1997) attempt to estimate the value of ecosystem services on a *macro level*: by drawing upon numerical findings in case studies and aggregating and extrapolating the derived figures across regions, the value of the world's ecosystem services is determined (Costanza et al. 1997). Because of methodological inconsistencies, this approach has invited criticism in the academic debate (Balmford et al. 2002; Turner et al. 2003).[34]

ments and modifications to this classification see, for example, Pagiola et al. (2005: 11) or de Groot et al. (2002). For a detailed methodological description of the approaches, see OECD (2002: 89ff.) or Kolstad (2000: 289ff.).

33 Nunes and van den Bergh (2001) and Turner et al. (2003) provide a critical review of these studies.

34 By addressing some of the shortcomings, Balmford et al. (2002) conduct a modified, bottom-up approach, without addressing the general caveats to this methodological approach (Turner 2003; Nunes and van den Bergh 2001).

Ecology and Economic Valuation

The concepts described for the valuation of ecosystem services, as well as the techniques for valuation in applied work, rely upon the assumption that values of biodiversity can be measured by individual *preferences*. In an alternative approach, the value of biodiversity is described using ecological *diversity measures*. Without referring to utility and preferences in the welfare economic sense, it is assumed that preserved diversity generates values in such a way that freedom of choice is maintained. By focusing on the value of species, evolutionary information given in phylogenetic trees and/or taxonomic information is used to describe the dissimilarities between species by making pairwise comparisons. By aggregating the information thus derived, numerical values can be assigned to subsets of species, which in turn facilitates their ordering (Weitzman 1992, 1993, 1998; Nehring and Puppe 2002, 2004).[35]

Although this approach relies upon ecological concepts, it represents an economic approach to valuation. It has to be distinguished from the approach in ecology. In this discipline, valuation is related to ecosystems and the extent of their functioning (ecosystem health/integrity) (Nunes et al. 2001).

With the described interaction between the ecological and economic system, there are calls for the greater *integration of ecological and economic knowledge* when valuing biodiversity from the human perspective. Instead of considering exogenous parameters of either the ecological or economic system, the analytical approach to the economic valuing of biodiversity is ideally fully coupled, i.e., a "unified" approach is used. For example, when it is assumed that decreasing diversity influences ecosystem stability and productivity, this has an impact on the market prices and social values of natural resources and thereby causes repercussions in natural resource management. Brock and Xepapadeas (2001) have been the first to make modeling efforts in this respect (Batabyal et al. 2003).

Finally, the described characteristics of ecosystem dynamics have implications for economic valuation. As argued above, valuation refers to resource management, leading only to marginal changes in ecosystem conditions (Fromm 2000; Hampicke 1999). It is implicitly assumed that these marginal changes are connected only with incremental changes in the flow of ecosystem services. However, due to the existence of ecological thresholds, the flow of ecosystem services may change abruptly in a nonlinear fashion (Limburg et al. 2002). This *nonlinearity in ecology* certainly influences the economic value, although the

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35 Several economic studies address the topic of valuation using ecological diversity measures, for example, Solow et al. (1993) or Moran et al. (1997). Mainwaring (2001) provides a detailed discussion of the differences between the valuation approaches and their underlying assumptions.

reaction in the value does not need to be nonlinear because of potential substitutions and changes in relative prices (Faber et al. 2002).

Aside from the existence of potential thresholds as such, valuation is affected by the fact that the location of a threshold and the magnitude of the impact caused by crossing the threshold is typically uncertain and sometimes cannot even be described in a stochastic framework. The viability of an economic valuation is further restricted if the crossing of the threshold and its associated impact on the ecological system are irreversible (Chavas 2000; Faber et al. 2002).

Limits to the Economic Valuation of Ecosystem Services

The concept of TEV suggests describing the contribution of biodiversity to human well-being by assessing the individual value components in monetary units and adding them to produce the total value. In this context, the literature highlights that the value of biodiversity is more than the sum of the values of its individual components. Valuable ecosystem services, particularly those that relate to ecosystem functioning, are generated due to *complementary relationships* between biological resources within and across different diversity levels (Fromm 2000). Considering species diversity, for example, a certain species may be crucial for ecological integrity and, therefore, for human well-being. This species' contribution often depends upon the sufficient population of another complementary species. Such values due to ecological complementary relationships are partly, albeit incompletely, considered in the indirect use value within TEV concept (Fromm 2000). Because of this incomplete assessment, TEV in its common definition understates the value of biodiversity.

A conceptual caveat to the valuation of ecosystem services relates to the understanding of monetary valuation in economics: generally, valuation refers to changes of goods and services on the margin. Marginal changes implicitly relate to the substitutability of different goods and services. Considering nonmarket ecosystem services, assigning monetary values to these goods and services makes them comparable to other nonenvironmental goods. This in turn may suggest that they can be substituted by other man-made services (Hampicke 1999). Nevertheless, findings in ecology show that ecosystem services that are crucial for the functioning of the ecosystem, can rarely be replaced by man-made services; they are *nonsubstitutable*. Accordingly, it would be inconsistent to assign monetary values to them. Consequently, whenever nonsubstitutable ecosystem services are involved, a monetary valuation is necessarily incomplete—independent of whether a methodologically proper approach is applied (Gowdy 1997; Hampicke 1999; Nunes and van den Bergh 2001).

Further difficulties in applied economic valuation stem from the fact that empirical evidence suggests that, for some people, the preferences for biodiversity conservation are *lexicographic* instead of utilitarian. This means these people are unwilling to substitute conservation for different goods, which imposes difficulties upon the aggregation of preferences (Hanley et al. 1995). Furthermore, people generally find it difficult to assess biodiversity when facing scientific uncertainty and ignorance regarding the importance of biodiversity for their well-being. Empirical evidence indicates that preferences for biodiversity conservation are rarely exogenously fixed but *sensitive to the information* provided to the people (Hanley et al. 1995).

To resume the debate, there is a consensus that, on the one hand, it is impossible to describe the importance of biodiversity and ecosystem services for human well-being in monetary terms due to their complex nature. Consequently, decisions about resource management that are critical for either the maintenance of or the decline in the flow of ecosystem services cannot rely purely on cost benefit considerations; on the other hand, it is recognized that a monetary valuation of marginal changes in an ecosystem is reasonable under many circumstances (Heal 2000; Gowdy 1997; Nunes et al. 2001).

Regarding the formulation of policies on resource management, imperfections in assigning monetary figures to biodiversity and ecosystem services should not draw attention away from implementing the correct incentives for its preservation. While incompletely described values from biodiversity affect the definition of objectives, the design of *incentive measures* is not necessarily dependent upon perfect information on values. From an optimistic perspective, incentive measures can provide additional information on the value of originally nonmarket ecosystem services. For example, by introducing market-based instruments for carbon dioxide abatement, prices may be assigned to ecosystem services that support carbon sequestration (Heal 2000).

2.3.3 Ecosystem Services and Resource Management

The previous two sections have dealt with the *information* needed for human decision making on resource management and ecosystem interference:

- The approach of an integrated ecological-economic system identifies the feasible options for ecosystem management and describes the flow of ecosystem services connected with possible individual ecosystem states.
- Applied methods of valuation reveal the contribution of (nonmarket) ecosystem services to human well-being.

Given the information in these two areas, decision making refers to the maximization of well-being under the constraints imposed by the resource endowment and the feedback effects resulting from the dynamic interactions within the economic-ecological system. A further constraint on the choice of the optimal management is imposed by *institutions* that regulate the interactions between the economic actors and that thereby affect the incentives for conserving and using individual biodiversity components (Heal and Small 2002).

Management decisions extend to both a spatial and temporal dimension. The power of decision is allocated among multiple actors. The stock and quality of biodiversity on an ecosystem level and between ecosystems results from the decentralized decisions of private, utility-maximizing households and firms acting as landowners or resource users. In addition, public sector authorities intervene in the private allocation by (1) regulating practices on land use and resource extraction and (2) implementing a management regime in areas that are state property. In order to control the allocation in this respect, the *property rights* regime on natural resources and ecosystem services is considered the core institution.

Biodiversity Uses and Degradation from an Economic Perspective

Economic analyses of resource management are conducted using either a positive or normative approach. A positive approach attempts to explain why owners of biodiversity conserve or deplete their resources under a given socioeconomic environment. The normative approach explores how conservation, which is desirable from the societal perspective, can be supported or, reversely and more commonly, how undesirable depletion can be prevented.

Regarding the *positive approach*, alternative but related concepts are used to explain why human use contributes to the degradation of valuable biodiversity (Barrett 1996):

First, the existence of *externalities*, or synonymously spillovers, refers to both the conservation and depletion of biodiversity. For conservation, it is evident that the resource owner who frequently appropriates only parts of the benefits from conservation efforts largely carries the costs of conservation. In other words, the beneficiaries of conservation participate insufficiently in the costs of conservation. Accordingly, the private net benefits of conservation are below the social benefits, with the consequence that the private resource owner underinvests in conservation. Alternatively, the private costs of resource depletion are below the social costs. This wedge favors the profitability of nonsustainable resource uses and, thereby, supports an excessive loss of biodiversity (Perrings and Opschoor 1994; Siebert 2005: 7ff., 206f.).

Second, a special case of biodiversity externalities is described by ecosystem services that are nonrival in consumption and whose consumption or use by third parties cannot be excluded (Cornes and Sandler 1996: 6f.). Ecosystem services that satisfy these conditions are *public goods* (see Table 1). Because of the public good properties, markets do not provide returns for private investments in resource conservation that generates ecosystem services of this kind. Moreover, when alternative resource use provides a positive income for the resource owner, the resulting land allocation decimates biodiversity to an undesirably low level (Heal 1995; Siebert 2005: 59ff.).

Third, biodiversity may represent a *common property resource* (Barrett 1996). The definition as a common property considers the property of nonexcludability from another perspective; it points to the method of resource management. The owners or claimants to biodiversity form a group of resource users who have to agree to an access and use regime (Stevenson 1991: 53ff.; Siebert 2005: 16ff.). In contrast to the concept of a public good, individual resource use diminishes the biodiversity quality that is available to other parties. An excessive depletion of biodiversity results when the agreed upon access and use regime is not able to induce sufficient investments in conservation or when it does not prevent over-exploitation by the individual group members (Barrett 1996; Lerch 1998).

By using these concepts, the problem of biodiversity loss can be framed on different spatial scales. While conservation generating external benefits is the result of decentralized decisions on local resource management, only some beneficiaries of conservation live in the proximity of that resource. Others reside at a distance, both on a national or international level. For example, global benefits of biodiversity primarily refer to the informational value of biodiversity in biotechnological R&D. Local benefits are predominately represented by the services that support the functioning of the ecosystem. While the former typically represent pure public goods, the latter are impure public or quasi-private goods (Perrings and Gadgil 2003).

Strategies for Conservation: Property Rights and Market Mechanism

As current biodiversity loss extends to a magnitude that is considered excessive and undesirable from a societal point of view, the concepts described imply that biodiversity loss is essentially driven by the discrepancy between private and social values of biodiversity (Perrings et al. 1995).

Moreover, in case of the identity of private and social values, decentralized transactions in *markets* for the various biodiversity components and ecosystem services are supposed to provide the appropriate incentives for the maintenance of biodiversity at an efficient and ideally socially optimal level. This is essentially the welfare economic paradigm saying that in the absence of imperfections, the market mechanism can provide an efficient allocation (OECD 2004: 23ff.).[36] The comparison between market allocation under imperfect conditions with socially optimal allocation as the benchmark is the basis for *normative analyses*. This approach deals with a *biodiversity policy* aimed at correcting existing market failures as far as possible.

An integral part of this policy is the specification and enforcement of well-defined *property rights*: individuals or communities vested with the right to control the access to and use of biodiversity are able to supply and exchange ecosystem services for reward. Thus, the right holders can capture the values of biodiversity, which in turn creates incentives to conserve and use biodiversity sustainably (OECD 2004: 49ff.). In contrast to this conceptual view, it is impossible to correct all forms of *market failure* in practice. Due to multiple externalities inherent to the nature of biodiversity and its uses, as well as the pure public good property of some ecosystem services, market mechanisms alone cannot support an efficient allocation. However, the market mechanism may encourage private conservation efforts (Heal 1995).

The tasks for biodiversity policies in this regard include the (1) correction of poorly defined property rights in case of open access regimes (e.g., Mendelsohn 1994; Swanson 1994) and the (2) definition of new property rights that support the creation of markets for ecosystem services for which they have not yet come into existence.

According to the possible types of ecosystem services described in Figure 1, *market creation* relies upon three channels:

- First, new markets emerge for products and services that are, in principle, rival and excludable in use but that embody a certain *private value beyond the resource-use value* (OECD 2003: 31ff.). Two types of goods are subsumed under this property: (1) natural resources for biodiversity-friendly consumption and (2) genetic resources for production and R&D.

  Regarding natural resources in private consumption that are cultivated and harvested in biodiversity-friendly conditions, separating the existing market into segments of differentiated products creates new markets. *Certification* and *labeling* schemes assist consumers in distinguishing conventional products from biodiversity-friendly ones. Examples are certified products of sustainable forestry or organic agriculture. The producers of these goods have an incentive to use resources in a conservation-compatible manner, since con-

36 The first and second theorem of welfare economics refer to the existence and establishment of Pareto-efficient outcomes in a competitive market economy. The existence of a social optimum implies that it is possible to aggregate preferences in such a way that just one of the potentially many efficient outcomes is preferred over the others.

sumers are willing to pay a premium for this sort of production. Policy intervention aims to support the process of differentiation through initiatives on information provision (Kotchen 2005; Nunes and Riyanto 2001).

The value of natural resources that are used in R&D is essentially determined by the *genetic information* embodied. By assigning property rights to these genetic resources and the information contained, the emerging markets ideally enable the owners to generate extra income, which in turn induces private incentives for conserving the habitat of these resources. Chapter 3 investigates the effectiveness of the property rights mechanism in capturing the value of biodiversity in this regard in more detail.

- Second, market institutions are introduced for ecosystem services that are nonrival in use but where excludability can be established so that the users of the services have to pay user charges. This relates to *club goods* such as (1) ecotourism and parks and (2) local ecosystem services.

  Regarding *ecotourism* and *parks*, access fees to scenic landscapes are charged in order to generate revenues that can be re-invested in the sustainable management of the conserved areas. Policies on property rights and market creation enact access provisions but possibly also the establishment of private-public partnerships for managing the conserved areas and enhancing the provided nonuse values (Huybers and Bennet 2003; OECD 2003: 53ff.). Chapter 4 reconsiders some aspects of ecotourism.

  Markets for *local ecosystem services* relate to indirect use values. Examples mainly refer to services in the context of watershed management. Private markets for these services do not develop spontaneously. Due to free-riding behavior and practical problems in enforcing exclusion from use, as well as a lack of awareness on the demand side, market creation typically requires substantial policy guidance. Accordingly, transactions are not characterized by spot market deals between private individuals, but rather by bilateral long-term contracts on compensation payments for suitable resource management that are concluded between municipalities and private landowners (Ferraro 2004; OECD 2003: 61f.).
- Finally, markets in the conventional definition cannot be created for ecosystem services that are virtually nonexcludable. The market mechanism can, however, support regulatory policies that aim at the provision and maintenance of services characterized as *pure public goods* or *open assess resources*. While restricting the total level of resource use to sustain the valuable flow of ecosystem services above a certain critical limit, the remaining use entitlements are assigned as individual property rights. The certified rights (permits) are then tradable among the interested parties. This type of cap-and-trade mechanism is applied, inter alia, to (1) fisheries and (2) land development (OECD 2003: 65ff.).

In *fisheries*, individual resource uses in marine ecosystems are restricted by quantity-based regulations. This is typically considered a form of command and control approach. In order to achieve efficient allocation, policy intervention does not restrict the permitted catch of the individual fisherman, but the total quantity permitted. Then, *individual transferable quotas* (ITQ) are assigned, whereas each fisherman can buy and sell quotas on a competitive market. In the ideal case, this regime guarantees the conservation of the resource stock within safe limits and promotes the efficient allocation of resources among the fishery enterprises (Newell et al. 2005; Siebert 2005: 132ff.).

In terrestrial ecosystems, the same mechanism is used to the regulation of *land development*. While resource use is restricted in quantitative terms in order to preserve vital ecosystem services, the landowners are entitled to *transferable development rights* (TDR), which they can sell on a market (Panayotou 1994). Extra land development in locations is dependent upon the possession of such titles so that the market demand for TDR is ensured and a market price for the preservation of natural land develops.

Policy intervention primarily refers to the establishment of a legal framework for land use and markets for TDR. Examples are schemes for wetland banking and carbon sequestration credits (Murtough et al. 2003). The existing regimes mostly extend to the regional level (Weber 2004). On a conceptual basis, regimes are also proposed for the continental level (Linden et al. 2004). Section 4.1 briefly reconsiders the cap-and-trade systems in protected area policies.

Supplementary Instruments of Environmental Policy

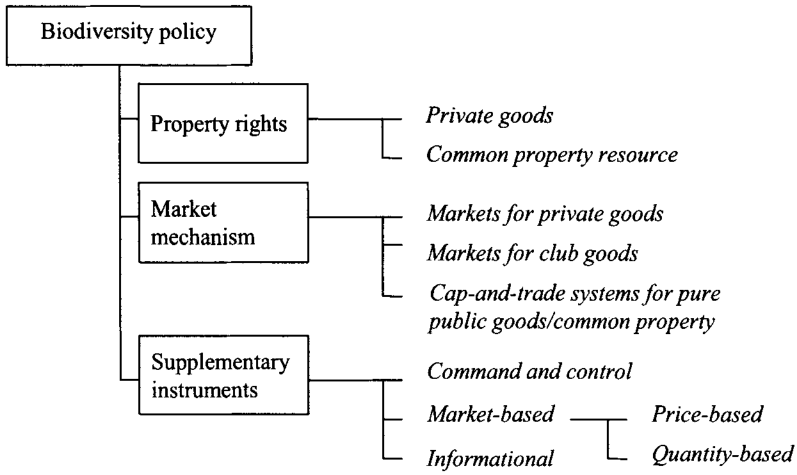
To assist policies on property rights and market creation, there are several instruments that can be used to influence the incentives for resource use. They either (1) change the *relative prices*, i.e., the private costs of conserving/depleting biodiversity or (2) increase the *income* of resource owners, depending upon the resource management they select (transfer policy) (Perrings and Opschoor 1994).

A conventional classification of these policy instruments first identifies *command and control instruments*, i.e., instruments that prohibit resource use that is detrimental to biodiversity. A widely discussed example in this regard is the US Endangered Species Act (Brown and Shogren 1998).

The second category refers to *market-based instruments*. These include (1) price-based instruments, such as fees, levies, and taxes that are charged for certain resource uses, subsidies, and income transfers paid by public authorities to reward private use assisting in the conservation of valuable but threatened biodiversity. In addition, (2) quantity-based instruments, such as the TDR mechanism, are subsumed under this category.

A third category—and a more recent set of instruments—relates to the provision of information on biodiversity and aims to increase the public awareness of

Figure 4:
Strategies for Biodiversity Conservation



biodiversity loss. The above-mentioned policies on (obligatory or voluntary) consumer information are examples of *informational instruments* (Nunes and Riyanto 2001; OECD 2003: 95ff.). Measures on public awareness aim to change the attitudes of stakeholders who are directly responsible for resource management. Besides providing information on, for example, sustainable production technologies, informational instruments attempt to influence the objectives of the economic actors by emphasizing the ethical values of biodiversity (moral suasion) (Siebert 2005: 131).

Figure 4 summarizes the classified approaches in biodiversity policy.

Policy Choices, Their Evaluation, and Policy Failures

Regarding the use of natural resources and ecosystems requiring policy intervention, several instruments, including combinations of individual instruments, can in principle be applied. In order to make a rational choice of the appropriate policy mix, the options available have to be evaluated against the background of selected criteria. The literature on environmental policy instruments names several criteria for the evaluation (Young 1992: 95f., 39ff.; Siebert 2005: 130f.). A traditional classification includes the following four criteria:

Environmental effectiveness, which means that the policy intervention reinforces ecosystem integrity: the flow of valuable ecosystem services is maintained and its quality is improved. Also, the policy implemented is capable of responding adequately to potentially abrupt changes to the ecosystem.

Economic efficiency, which means that no other intervention can support an allocation of resources that leads to significant welfare improvement without reducing the utility for some individuals (Pareto improvement). Furthermore, set environmental targets are reached with minimum costs. The intervention encourages technological improvements in resource management, leading to an improved environmental quality (dynamic incentives).

Equity and acceptance, which means that intervention is politically accepted because it does not give rise to, but rather mitigates, social conflicts about the distribution of economic resources, rights, and wealth. Acceptance requires the stakeholders to consider the allocation resulting from the intervention as essentially fair.[37]

Feasibility, which means that the intervention is enforceable at the appropriate costs. In this regard, information requirements are not excessive. Uncertainty regarding ecological responses does not impede the achievement of the policy target.

Generally, the evaluation of alternative, feasible policy mixes assumes a rational decision-making process in order to maintain valuable biodiversity components. In practice, however, decisions are subject to imperfections in the political processes resulting from misrepresentations of preferences (lobbying, corruption, political discrimination) and/or a lack of public awareness on the issue of biodiversity loss. When governance capacities fail in these two respects, interventions in ecosystem management are not based upon rationality and may create perverse incentives for resource depletion and the nonsustainable use of biodiversity. In practice, such *policy failure* is often a problem in developing countries. It becomes visible, for example, in the failure to specify and enforce property rights that control the extraction of natural resources or in subsidies for resource-based industries that promote the resources' overexploitation (Smith et al. 2003).

The existence of policy failure provokes calls for the correction of the policy intervention, which creates such perverse incentives for biodiversity conservation. In practice, social conflicts and powerful interest groups in political processes frequently impede a policy change towards a rational design of interventions (OECD 2004: 115ff.).

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37 It is sometimes argued that equity and efficiency should play a role of equal rank (Young 1992: 39ff.). While the market mechanism primarily intends to provide economic efficiency (regardless of distributional effects), there has been a call for additional instruments to be used to attain an equitable distribution of benefits generated in a market allocation (Perrings and Gadgil 2003). Interactions between efficiency and equity are investigated in sections 3.1 and 4.3.

2.3.4 Temporal and Spatial Aspects of Resource Management

Biodiversity is related to particularities on the temporal and spatial scale that I have not yet addressed in the description of ecosystem services and resource management.

Human Use and Conservation Policies in a Temporal Dimension

When considering first the temporal dimension of the human use of biodiversity and efforts towards its conservation, the academic and political debate addresses mainly two issues, which are partly interrelated. These are (1) sustainability and (2) irreversibility of resource management.

As far as *sustainability* is concerned, there are multiple ways of operating this concept. From a political perspective, sustainability is related to the question of intergenerational equity. More precisely, it is asked whether the current institutions can arrange for the use of biodiversity that supports the wellbeing of present generations "without compromising the ability of future generations to meet their needs" (Brundtland report) (Heal 1998: 1ff.). The debate calls for a definition of sustainability in an ecological, economic, and social dimension. From a narrow economic perspective, the major issue is to the optimal, long-term resource/ecosystem management.

To answer this normative question, assumptions on the preferences of future generations and how they should be weighted relative to the preferences of the present generation are required. The decision making in this respect depends upon ethical beliefs rather than economic cost benefit considerations (Barrett 1992; Pearce and Opschoor 1994). Accordingly, economic research on sustainability concentrates on positive analyses that assist decision making by describing how different economic drivers influence the long-term preservation and use of biodiversity.

Analytical studies in this regard typically rely upon neoclassical growth models with natural resources as inputs in production, whereas preserved biodiversity generates positive stock externalities. Given an initial resource endowment, the models search for the extraction path that maximizes the discounted flow of social utility. It is then investigated how changes in the socioeconomic environment influence the optimal path and long-term equilibrium of resource preservation/depletion. Such changes are stylized, for example, by alternative assumptions on the time preference (discount rate), technical progress, population growth, or resource endowment (Barrett 1992; Fisher et al. 1972; Krautkraemer 1985; Li and Löfgren 1998; Rowthorn and Brown 1999).[38]

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[38] These studies differ with respect to technological transformation possibilities, as well as with respect to the nonuse values of biodiversity.

Using the analytically derived path of extraction, i.e., the prediction of ecosystem change, it is generally asked whether the process of (resource-based) development is self-correcting under certain circumstances. The question is whether habitat destruction comes to a halt at some level, at some stage of development, and whether the biodiversity loss incurred is at best reversed in the further course. If such a favorable trend of change were indeed observed, this would support the hypothesis of an *environmental Kuznets curve* (EKC). In this case, increasing resource-based development contributes to increasing income/ consumption per capita that in turn decreases the value of the extraction-based goods relative to long-term preservation. Therefore, prospering countries increase investment in preservation, which in the long run supports the restoration of biodiversity.

Empirical investigations of the relationship between growth and biodiversity conservation are generally constrained by difficulties in measuring biodiversity, the uncertainty involved, and the fact that biodiversity loss occurs due to the complex, often site-specific synergy of the multiple drivers. Empirical findings indicate that the EKC hypothesis, in fact, does not apply to biodiversity (Asafu-Adjaye 2003; Dietz and Adger 2003; Barbier and Burgess 2001). Consequently, the forces of economic development are not believed to secure the conservation of global biodiversity.

Two aspects of this nexus have attracted further attention: first, the studies on the growth biodiversity relationship typically take a macro perspective and, per se, assume a *trade-off between development and preservation*. This assumption is challenged by other studies that consider biodiversity conservation and resource use from a micro perspective: assuming a positive relationship between growth and poverty reduction, it is asserted that, sometimes, conservation is only achieved when the interests of poor populations residing in or next to biodiverse habitats are taken into account. In other words, successful activities demand the integration of development and conservation interests (Perrings and Gadgil 2003; Swanson and Kontoleon 2000; see Section 2.2). If this is true, the interest shifts from the optimal extraction path to the search for appropriate measures that enable conformity between the development and conservation goals.

The second feature in the temporal dimension is *irreversibility* in biodiversity degradation. As explained above, irreversibility refers to the existence of thresholds and ecological feedbacks on a local scale (Dasgupta and Mäler 2004).[39]

39 On a macro level, irreversibility does not exist as such but may be approximated by aggregating the multiple irreversible ecosystem changes. In this way, irreversibility fundamentally questions the pattern of an EKC: once a critical limit in degradation is passed, no restoration is possible, no matter how the development process proceeds and the relative value of preservation evolves.

Irreversibility in preservation is typically connected with uncertainty.[40] Two sorts of uncertainty can be identified: first, by exceeding the threshold, unforeseeable ecosystem changes may occur. Second, the level at which resource use triggers irreversible and undesirable ecosystem change is uncertain (Perrings and Pearce 1994). The nature of uncertainty is described either as (1) an inherent risk to economic activity or (2) a scientific uncertainty that may become clear over time as people learn about ecological relationships in the future but face a decision on present resource use. Against this background, analytical studies focus upon how the existence of irreversibility influences management decisions on present resource use and the role of *learning* in this regard (Immordino 2003).

Two classical studies by Arrow and Fisher (1974) and Henry (1974) imply that, with the possibility of obtaining better information, it is rational to restrict present irreversible resource development to keep future options open (irreversibility effect). Preservation in this regard creates a *quasi-option value* (Fisher and Hahnemann 1987). The subsequent literature has qualified this finding and refined the conditions for the validity of the irreversibility effect (e.g., Ulph and Ulph 1997; Immordino 2003).

In order to attain preservation, certain use restrictions have to be imposed. A combination of measures such as quantity-based regulation and cap-and-trade mechanisms are typically applied in this regard. Since discontinuity combined with uncertainty impedes the application of conventional cost-benefit calculus, use restriction as a result of standard setting is based virtually on ecological grounds and ethical judgments on the social acceptance of such a restriction (Perrings and Pearce 1994).[41]

Regulation in this respect is defined by a set of *safe minimum standards* (SMS) as a realization of the *precautionary principle*. The SMS approach is based conceptually upon the rudimentary mini-max decision rule, which states that, given alternative outcomes, a standard should be set in order to minimize the maximum welfare loss that can occur due to various resource uses. An exception to this rule should only be allowed if the costs of the standard are un-

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40 Barrett (1993) considers a modeling of irreversibility with perfect information. It turns out that it is rational to preserve biodiversity above the levels that refer to the initial benefits and costs of preservation. This is because marginal benefits of biodiversity relative to the marginal cost of conservation may rise over time but, due to its irreversible loss, it makes it more difficult to meet the future needs for biodiversity.

41 Perrings and Pearce (1994) argue that safe minimum standards do not necessarily imply a command and control policy. Market-based approaches can also be fruitful if a policy intervention can guarantee that the private cost function of resource use reflects the discontinuity and uncertainty of ecosystem changes. A means in this regard may be credible penalty schemes that arranges for private costs equal to social costs at the threshold (Perrings and Opschoor 1994).

acceptably high (Bishop 1978; Ciriacy-Wantrup 1952: 251ff.; Crowards 1998). Because of its simplicity, the decision rule can also be applied even when scientific information is insufficient to identify probabilities and assign them to potential outcomes.

Human Use and Conservation Policies in an International Dimension

Regarding biodiversity use and conservation on a spatial scale, the beneficiaries of conserved biodiversity frequently are not the providers of biodiversity or do not even live in their proximity. In order to create incentives for conservation, mechanisms are needed that facilitate the exchange of biodiversity values over long distances. Depending upon the private or public good nature of the ecosystem services, these mechanisms are either (1) international trade with natural resources or (2) international transfer schemes that provide for compensation payments for local conservation efforts.

With respect to *international trade in natural resources* that are extracted from biodiverse habitats, the public focus typically falls on trade in wildlife and tropical timber. Furthermore, the international exchange of genetic resources attracts some attention. In addition, the world market supply of ecotourism services is considered a specific variety of international trade in biodiversity (Heal 2003).

The extensive literature on the environment-trade nexus typically concentrates on the trade-pollution linkage (Siebert 1996, 1977). Only recently has attention been drawn to the relationship between trade and renewable resource management (Barbier and Bulte 2004; Bulte and Barbier 2005).[42] In the academic discussion, as well as the political debate, the focus falls upon the impact of policies that either stimulate *trade liberalization* or support *trade regulation*. From a descriptive perspective, it is asked how the opening up of an economy to international trade influences the allocation of biodiversity within and among countries (environmental effectiveness). The normative approach considers how trade and its potential regulation affects human well-being (efficiency).

In analytical studies on trade in wildlife, biodiversity is typically approximated by a specific species, for example, the elephant, which is represented by a homogenous renewable resource stock (Bulte and van Kooten 1999).[43]

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42 Generally, there are similarities between biodiversity and pollution in the context of trade, namely international environmental spillovers, the role of trade interventions, and the relationship between national regulation and competitiveness on international markets. Otherwise, the properties of renewable resources and ecosystems that I have described above make biodiversity quite distinctive from the pollution issue (Barbier and Bulte 2004; Bulte and Barbier 2005).

43 Kremer and Morcom (2000) consider the case where natural resources (elephants) are used to produce a storable good (ivory). This resource characteristic offers room

Considering firstly a *partial equilibrium framework*, trade induces a country rich in biodiversity to become a net exporter and causes the domestic resource price to increase in this context. For given extraction costs, trade therefore creates an economic rent for the exporting country, which in turn increases the incentive for resource extraction. If, in this regard, the resource stock demonstrates a low regenerative capacity and the time preference is relatively high, the trade-induced price increase may contribute to the long-term depletion of the resource stock, i.e., the extinction of wildlife. Through a rise in the resource price, the surplus of the domestic supplier is increased more than the surplus of domestic consumers is reduced, so that the welfare of the resource country is improved. This conclusion rests upon the assumption that domestic producers compensate the consumers and that wildlife does not embody any nonuse values (Burgess 1993; Bulte and Barbier 2005).

When such nonuse values are present, trade regulation is interpreted as a means to make the resource owners internalize the social costs of extraction. Through the use of tariffs or even trade bans on wildlife products in the importing country, the resource price on world markets decreases. In the ideal scenario, this relieves some of the pressure on domestic stocks in the exporting countries and preserves nonuse values (Barbier and Rauscher 1994).

The partial bioeconomic framework does not take into account the fact that wildlife habitats may be productive in *alternative land use* (Barbier and Bulte 2004). In this respect, biodiversity is not only threatened by excessive resource extraction but also by habitat conversion for alternative production. However, extraction does not have to be excessive. Moreover, opening up to international trade may generate private returns from world markets for wildlife products and thereby create private incentives for sustainably managing the resource stock. In this regard, trade actually stimulates long-term conservation. Since trade regulation typically decreases the domestic price of wildlife products compared to the price of goods from other land use, trade interventions with respect wildlife products can be counterproductive for conservation (Swanson 1994, 1995a).

When commodities generated from converting land use are also exchanged on world markets, trade influences the incentive for sustainable resource management and habitat preservation in different ways. The impact of trade liberalization or regulation is studied in the *two-commodity* framework of a *small country* (Barbier and Schulz 1997; Barbier and Rauscher 1994). It is shown that if a biodiversity-abundant country opens up to international trade, it may increase resource extraction because of an increasing domestic resource price. At the same time, the country may reduce habitat conversion, since alternative consumption

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for strategic behavior in wildlife trade, both for the suppliers and the regulatory authorities (Bulte et al. 2003).

goods do not need to be produced on converted domestic areas but, rather, can be purchased from abroad. Biodiversity is described by the resource stock, whose size does not only depend upon the usual regeneration dynamics but also the size of the surrounding habitats. Regarding the change in the preserved resource stock after trade liberalization/regulation, it cannot be determined unambiguously whether opening up to trade favors or weakens the incentive to conserve. The overall impact depends upon the specific terms of trade and the effect of substitutions among the involved commodities in domestic consumption (Barbier and Schulz 1997; Barbier and Rauscher 1994).

Generally, trade in conventional commodities leads to economic *specialization* according to comparative advantages (Siebert 2000). With regard to trade in biological resources, it is asked in the literature whether specialization within a country that opens up to trade changes the management of ecosystems in such a way that biodiversity declines both on a national and global level (homogenization) (Gale 2000). Polasky et al. (2004) use a general equilibrium model with two tradable natural resources. The model accounts for differences in the species' assemblage between two countries and natural areas within each country. Trade then leads to specialization within one country, i.e., the country's natural resource production shifts to one type of natural area. The species richness in habitats of this type decreases, while the species richness in the other natural areas regenerates itself. Depending upon the species productivity in areas of different habitat classes, trade leads to a decline in national biodiversity relative to a situation under autarky.[44] When global biodiversity is defined as the sum of the national endowments of species diversity, the overall impact of trade upon biodiversity depends upon the overlap of species diversity among countries. It turns out that strong asymmetry in the species assemblage (endemism) leads to an overall decline in global biodiversity.

The previous approaches implicitly assume *optimal property rights* on land and natural resources, i.e., an optimal management regime can be implemented and monitored. However, many real world situations are characterized by *imperfections* in property rights enforcement, a de facto open access regime, and illegal resource extraction (wildlife poaching) (Bulte and Barbier 2005; Barbier and Bulte 2004). Several studies investigate the impact of trade policy on such a second-best world. Regarding the partial framework of an open access resource (with and without habitat conversion), it is shown that overexploitation occurs but that, depending upon the resource price/demand, a positive resource stock

44 This finding rests upon the assumption that species richness in natural areas of two alternative types is described by habitat-specific, concave species area relationships. Because of this concavity, the decline in species richness in the intensively used habitats is not offset by regeneration in the less intensively used habitats of the other type (Polasky et al. 2004).

may be preserved in the long run. It turns out that opening to trade affects the resource stock in the same way as under an optimal management regime. However, in the case of open access, trade does not change the welfare of the resource country (Barbier and Strand 1998; Bulte and Barbier 2005; Bulte and van Kooten 1999).

Turning to the general equilibrium framework with a renewable open access resource, the literature finds different and sometimes even opposing results for the impact of international trade.[45] Under certain circumstances, free trade may not only lead to resource overexploitation but also worsens the welfare of the exporting country. In the absence of effective policies on property rights, trade regulation represents the second-best instrument (Brander and Taylor 1998; Chilinisky 1994; Karp et al. 2001). This finding is qualified when alternative land uses for wildlife habitats are taken into account (Smulders et al. 2004).

Another strain of literature assumes *endogenous property rights* on natural resources that are determined by the institutional environment (Barbier and Bulte 2004): resource owners enforce their rights to natural resources, depending upon the benefits and costs of doing so (Eggertsson 1990: 83ff.). For example, in autarky, illegal extraction may be tolerated initially but if trade liberalization enhances the private resource value and therefore changes the benefit to cost ratio of property rights enforcement, a regime change from open access to protected property rights is induced. Since this change would also restrict non-sustainable resource extraction, trade in this respect brings about a positive impact on conservation. However, the impact on welfare is ambiguous since a stricter enforcement decreases the poaching labor force and hence increases the labor market supply, which in turn leads to a decline in labor income (Hotte et al. 2000).[46]

Turning to *international transfers*, such policies pursue an alternative approach to internalize biodiversity spillovers. Unlike with national spillover effects, there is no authority on an international level that can enforce internalization within countries. Countries are sovereign in their decision as to how to use their natural resource endowments. Accordingly, any extra conservation activities beyond the domestically determined level have to be agreed upon in a voluntary collaboration between the resource country and the international community. In order to make the resource country participate, it has to be compensated for the costs incurred by the extra activities. For this purpose, (financial) transfers are provided.

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45 These studies are surveyed in Bulte and Barbier (2005).

46 Trade can have an unintended impact upon biodiversity by supporting biological invasions. Costello and McAusland (2003) and McAusland and Costello (2004) analyze the role of trade policies in this regard.

In practice, transfer donors are represented by *multilateral organizations*, such as the Global Environment Facility, or individual countries that offer payments on an *official bilateral* basis. In addition, the private sector also provides some money in compensation for conservation arrangements. A well-known institution in this context is *debt-for-nature swaps* (e.g., Deacon and Murphy 1997). The practical tasks for international coordination concern the normative aspects of managing domestic resources in a way that global interests in them are protected. On an operational level, this includes an agreement on the equitable sharing of the benefits arising from such management, as well as on the sharing of the management costs (Amelung 1993; Cervigni 1998).

The literature on international transfers for biodiversity conservation typically assumes a neoclassical world. Transfer payments for conservation are then believed to unambiguously increase the level of long-term conservation and, furthermore, contribute to a global Pareto improvement. For payments on a country level, this is because the transfer relaxes the economic constraints of a biodiversity-abundant country and thereby allows that country to reallocate its resources for other purposes (Cervigni 1998; Barbier and Rauscher 1997; Barbier and Schulz 1997).[47]

A further focus of both descriptive and normative studies of transfer policies is the analysis of *free riding among donors*. International transfers are typically provided for the conservation of biodiversity that generates ecosystem services, which are nonexcludable and nonrival in their use. Services with alternative properties are traded on world markets. Accordingly, the raising of funds for biodiversity as global public goods within the international community suffers from strategic behavior among the individual donors. To investigate this issue, the literature relies upon the theory of private public good provision in public economics (Sandler 1993), as well as the game-theoretic considerations about enforceable, international environmental agreements (Barrett 1994.). Sections 4.1 and 4.4 provide a detailed description of these issues.

Transfers are sometimes perceived as transactions on an international market for land use rights: international donors "rent" areas in developing countries that are preserved or managed in a way that contributes to conservation and thereby to the generation of globally beneficial ecosystem services (Pearce 1998: 194ff.; Swanson 1995a). Nevertheless, transfer policies display several distinctive features compared to market transactions with renewable resources. Aside from the before-mentioned differences in economic properties, market-based conservation typically refers to private ventures. In contrast, transfer policy is mainly an official

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[47] Muller and Albers (2004) consider the micro level of transfer payments. By abstracting from the generation of funds for transfers on a national and international level, the authors investigate how to design compensation payments to rural people in developing countries.

governmental venture. Furthermore, the policy is backed by international agreements that result from coordination outside the market. Accordingly, it seems justified to distinguish the market-based conservation strategies from a collective approach to preserve public good-like biodiversity.

Synthesis

The key economic issues unfolded in Section 2.3 can be related to the findings on biodiversity presented in Section 2.2. In this context, the Convention of Biological Diversity (CBD) represents the outcome of international coordination. It can be interpreted as an internationally agreed upon regime for resource management. In its provisions, several of the aspects and instruments mentioned are addressed. According to Pearce and Perrings (1995), the CBD primarily emphasizes two instruments:

Property rights and market creation for genetic resources and genetic information. This measure is implied, inter alia, by the CBD's objective of "a fair and equitable sharing of the benefits arising out of the utilization of genetic resources" (Art. 1 CBD) in connection with the assignment of property rights to the individual countries (Art. 15 CBD).

Transfers to the developing countries that compensate for the incremental cost of biodiversity conservation in the interest of the international community. The CBD recognizes the global disparity in endowments of valuable biodiversity in favor of the developing countries and calls upon the developed countries to support the former financially in their efforts towards biodiversity conservation (Art. 20 CBD). Since signatory countries are called to "establish a system of protected areas or areas where special measures need to be taken to conserve biological diversity" in Art. 8(a) CBD, international financial support is partly used for this purpose.

In the following two chapters, I investigate these two major topics in more detail. Chapter 3 begins with an analysis of commercial use of and trade in genetic resources and market-induced incentives for biodiversity conservation.

3 Market-Based Incentives to Preserve Biodiversity: Commercial Use of, and Trade in, Genetic Resources

3.1 Genetic Resources as Economic Goods

Genetic resources are considered goods that embody a "private value beyond the simple and immediate resource-use value" (OECD 2003a: 31). In contrast to other biological resources, commercial use does not focus upon the material itself but rather upon the genetic information it contains (Small 1998: 1).

This section describes the properties of genetic resources and their commercial use (Section 3.1.1) and, related to this, summarizes the major aspects of the property rights issue with respect to genetic resources (Section 3.1.2). In order to define the question of further research, Section 3.2 continues with a discussion of the relationship between the use of genetic resources and biodiversity conservation.

3.1.1 Economic Properties and Commercial Use

For a study of genetic resources, these goods have to be distinguished from other potentially similar goods. In order to describe genetic resources, a commonly used definition and classification is given in Art. 2 of the Convention of Biological Diversity (CBD). According to this definition, a *genetic resource*

- represents "genetic material,"
- that is of "actual or potential value."

Genetic material, in turn, is defined in Art. 2 of the CBD as "material of plant, animal, microbial or other origin containing functional units of heredity."

According to this CBD terminology, genetic resources are perceived as a subset of genetic materials (OECD 2003b). However, since it is difficult to demonstrate that some genetic material lacks any potential value, the term "genetic material" is often used synonymously to mean genetic resources. Furthermore, the distinction between material and resources depends largely upon its actual human use and not upon the material as such (Byström et al. 1999).

The CBD introduces *biological resources* as a third good in this context. According to the CBD's provisions, biological resources "include genetic resources, organisms or parts thereof, populations or any other biotic component of ecosystems with actual or potential use or value for humanity" (CBD Art. 2).[48] The wording implies that genetic resources represent a subset of *biological resources* (OECD 2003b). Furthermore, since the definitions of genetic resources and biological resources are based upon the same criterion, there is apparently a considerable overlap between the two concepts. Accordingly, both terms are frequently used as synonyms (Allem 2001). In order to separate biological resources from genetic resources, Byström et al. (1999) argue that the focus should fall upon the hereditary units. Generally, these units have two functions: (1) the controlling of the biochemical processes in a cell and (2) the transfer of information to the next generation (Wolfrum et al. 2001: 35).

Units of Heredity and Values

The functional units of heredity contained in a genetic resource represent the informational content and are therefore essential for the resource's value. Given the multiple values of biodiversity, the CBD definition does not restrict the types of values that apply to a genetic resource. Moreover, in its preamble, the CBD specifies the values of biodiversity as "ecological, genetic, social, economic, scientific, educational, cultural, recreational, and aesthetic values." In addition to these terms, which refer to *utilitarian values* and are also captured in the concept of the total economic value, the preamble explicitly names the "intrinsic value" of biodiversity as a further category. Nevertheless, considering the assumed close relationship between the genetic and biological resources and the reference to the "value for humanity" in the definition of the latter, it can reasonably be argued that, without ignoring the intrinsic aspects, the value of genetic resources relates to utilitarian values in the first place (Allem 2000; Byström et al. 1999). Furthermore, regarding the utilitarian values, it is typically assumed that, although genetic resources can in principle yield nonuse values, such as bequest and existence values, *use values* in the form of direct use values and option values may dominate the total economic value of genetic resources.

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48 Outside of the CBD, the term *natural resources* is used. According to the understanding in economics, natural resources represent goods that are provided by nature to directly or indirectly satisfy human needs. This term includes factors of production, as well as consumption goods (Siebert 1983: 2f.). Since natural resources include (nonrenewable) mineral resources, as well as (renewable and nonrenewable) gaseous resources, biological resources represent only a subset of natural resources.

Classifying Genetic Resources

In general, genetic resources display a varying nature, depending upon their specific economic use. To describe them in more detail, these uses can be categorized according to different criteria. Possible classifications include categorization with regard to:

- the biological taxon , i.e., the reference to the biological origin,
- the surroundings, from whence genetic resources are extracted or in which they are conserved,
- the different types of human use.

Regarding *biological taxa*, genetic resources can be subsumed under plant genetic resources (PGR), animal genetic resources (AnGR), and microbial and other materials. Strictly speaking, one can define human genetic resources as a further subcategory. However, in political and scientific discussions, the issues of biodiversity are usually separated from the issues of human genetic resources (ten Kate and Laird 1999: 45).

Furthermore, genetic resources can be categorized according to the conditions in which the material can be found and is conserved. In this regard, *in situ* conditions are distinguished from *ex situ* conditions. The CBD (Art. 2) defines the former as "conditions where genetic resources exist within ecosystems and natural habitats and, in the case of domesticated or cultivated species, in the surroundings where they have developed their distinctive properties." In contrast, conservation in ex situ conditions means "the conservation of components of biological diversity outside their natural habitats." The distinction according to this ecological criterion carries economic implications, as remainder of the study will illustrate. The issues in this regard are, inter alia, the substitution of in situ material with ex situ material and the (evolutionary) changes in the stock of materials over time.

Finally, genetic resources can be categorized with respect to their human use, which according to the definition of genetic resources, relates to the use of the functional units of heredity and their expression in specific properties and qualities (Byström et al. 1999). On the one hand, this includes the *local and indigenous use* of the specific qualities of particular species, for example, for the purpose of medicinal treatment or seed production. This use is usually characterized by a relatively low degree of human modification and processing. Today, such use is of particular importance in developing countries.

On the other hand, genetic resources enter into *commercial and scientific use*, i.e., processes of biotechnological research and development (R&D), whereby the capital-intensive R&D technologies are predominately located—or at least constructed—in the developed countries, while the genetic materials used are to a significant part acquired from the developed countries.

This study draws the attention to commercial use. This is because, against the background of the issues of global biodiversity loss and the North-South conflict of interests as motivated in the previous chapter, I believe that it would be fruitful to concentrate upon commercial use, since it is of importance on a local, national, and international level of biodiversity policy. At the same time, I emphasize that this perspective does not ignore the importance of indigenous economic use on a local level.

In the literature, the commercial use of genetic resources is classified according to industrial sectors (ten Kate and Laird 2000) and/or product categories (Artuso 2002). These sectors and categories can be further summarized in three major areas, namely health care, agriculture, and other biotechnological production (Hill 1999).

- The *health care* area includes the production of pharmaceuticals and botanical medicine.[49] Natural products for personal care and cosmetics are also included in this area.
- Commercial use in the *agriculture* area is, in essence, described by the production of seeds for crops. The literature adds the biological products of crop protection and horticulture.
- The area of *other biotechnological production* is described by biotechnological applications with genetic resources that carry out (1) biotransformations for the removal of waste or pollutants or that serve the (2) material production, (3) energy production, or (4) other production of food.

The term biotechnology typically does not refer to a clearly defined industrial sector. In a broader sense, biotechnology includes all uses of living organisms for the purpose of producing or modifying goods, breeding useful plants or animals, and/or the development of microorganisms for special functions in biological-technical processes.

This perspective includes both traditional uses and modern capital-intensive applications. In an alternative and narrower definition, biotechnology refers to the industrial use of recombined deoxyribonucleic acid (rDNA), cell fusions, and new bioprocess techniques (OTA 1991: 5). Applications that fall within the narrow definition are frequently referred to as "new biotechnology" and are

[49] Botanical medicine refers to medicinal products of plant origin that are used in a crude or processed form. In contrast to pharmaceuticals, the use of botanical medicine does not include the isolation of single genetic compounds. Frequently, the term phytomedicine is used synonymously to describe botanical medicine, although it refers more to products based upon herbs, while botanical medicine may include nonherbal ingredients (EC-CHM 2005).

currently drawing considerable public attention. In the reminder of this study, productions are based upon conventional and new biotechnology are included in the sectoral representation of pharmaceutical production and seed production or they are summarized under other biotechnological production.

Different combinations of characteristics in the alternative categorizations described help identify specific subclasses of genetic resources. This is particularly true for the genetic resources of plant origin that are used in the agricultural sector. These resources are defined as plant genetic resources for food and agriculture (PGRFA) (FAO 1998: 13). While PGRFA are mainly ex situ genetic resources, they also encompass in situ plant genetic resources.

Genetic Resources: Information Goods versus Conventional Natural Resources

Since the units of heredity contained in a specific genetic resource determine its resource value, they have been at the center of economic analysis. From economic perspective, functional units of heredity are typically considered information. Information goods, in turn, demonstrate the following general properties (Varian 1999):

- Users must experience the information good before they know what benefit it provides them. Thus, information is an experience good.
- The production of (new) information is typically connected with a high fixed cost of production but a low marginal cost of reproduction, i.e., the production is characterized by increasing returns to scale.
- The use of information goods is typically nonrival, i.e., no user suffers any loss in utility if the same information is shared with other users. In addition, information is sometimes nonexcludable, i.e., information sometimes represents a pure public good.

It can be illustrated that each of these properties applies to genetic resources in a specific way. The usefulness and, hence, the economic value of a genetic resource is only revealed after it has undergone scientific assessment and testing (Allem 2000). This process usually comprises several stages, whereas the further prospecting of a resource only continues if some promising information has been found in the previous stages (Artuso 1996a). While the value of an examined genetic resource is made apparent step by step, it is initially *uncertain* from the point of view of the researchers.

Two things have to be considered with respect to uncertainty. First, when the value of specific genetic material is uncertain ex ante and the genetic material is transferred from the original owner to a researcher who carries out further testing, the risk of an unsuccessful research outcome can be shared between the two (Artuso 1996b; Sedjo and Simpson 1995). Second, by screening the material, information on the properties of screened resources is generated. This information is a public good and can be useful to other researchers who test the same genetic resource in a similar or wholly unrelated research process (Small 1998: 81ff.). In effect, information concerning a specific genetic resource may reduce ex ante uncertainty. In practice, traditional and/or indigenous knowledge of specific genetic resources often serves as an indicator of promising information (ten Kate and Laird 1999: 60f.). Consequently, specific knowledge serves to reduce uncertainty.

Following the definition of the functional units of heredity, genetic information can easily be replicated by *reproducing* genetic material as the carrier of information in the natural, biological way. If there are no specific barriers to reproduction, the marginal cost of reproduction is quite low. As the analysis in this chapter will show, this property applies to many commercial uses. It implies that users of genetic resources can appropriate the value of genetic information without the need of additional material from the original provider.

However, it is evident that, in some cases, there are (temporal) constraints on the technical *feasibility of replication*, for example, genetic information observed in a specific natural product can neither be rebuilt on a synthetic basis nor replicated using biologically related materials (Day and Frisvold 1993). In this case, the use of information requires the possession of the original carrying unit, i.e., the genetic material of specific taxa extracted from the in situ environment. Accordingly, these types of genetic resources no longer present information goods.

A further difference to other information goods, such as products in the software or music industries, is that genetic information is not necessarily created by human efforts. In addition to human efforts in R&D, evolutionary adjustments to nature guarantee that new genetic information is created. The use of naturally occurring genetic information in the R&D-based creation of information has important implications for the granting of intellectual property rights (IPRs), as the next section will show.

The production of wild but unimproved genetic information is not connected to fixed costs. The opposite result may apply to genetic information that is processed and improved in a capital-intensive R&D process. In addition, under certain circumstances, the marginal cost of reproduction is rather large due to biochemical constraints upon reproduction.

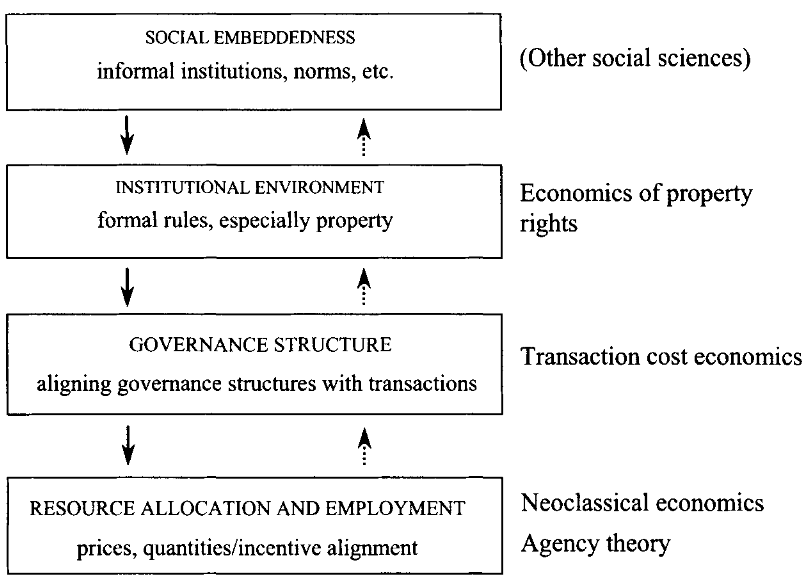
Furthermore, as with other information goods, genetic information is characterized by *nonrivalry*: no user, provided that genetic material of the same constitution is available in sufficient quantities, is disturbed by any other user of the same information. Depending upon the specification of use rights, genetic information represents an impure public good. The availability requirement highlights the fact that nonrivalry in the use of genetic information has to be separated from rivalry in the use of the resource as its carrying unit.

A genetic resource somehow represents a bundle of two distinctive goods: the material and the information. Someone else cannot simultaneously use the same genetic material that is in the possession of one researcher. However, the same genetic information may be contained in many samples of biological material of the same genetic constitution. Several researchers can be in possession of these samples at the same time (Sedjo 1992). Thus, the exclusive use of genetic material does not necessarily imply exclusive possession and use of the information it contains.

Genetic Resources in a Framework of Institutions

Given the definition of genetic resources and a brief description of their commercial use, genetic resources can in principle be studied on various but interconnected *analysis levels*. Each level emphasizes different *institutions* and generally requires different methodological approaches. The level of analysis can be classified according to the classification by Williamson (1998) described in Figure 5.

Figure 5:
Genetic Resources in a Framework of Socioeconomic Institutions



Source: Williamson (1998), own representation.

Customs, traditions, religion, social norms, or other informal institutions describe the top level (*social embeddedness*). Several social sciences deal with the analysis of these issues. In economics, the parameters at this level are typically considered as given. The second level in the figure concerns the *institutional environment*, i.e., the formal rules regarding human activities. This encompasses all humanly devised constraints that aim at structuring political, social, and economic interactions (North 1991). The analysis of these aspects relies primarily upon the economics of property rights and addresses normative questions on the proper design of the institutional environment.

Given this environment as a term of reference, the next level considers how the organizational structure of economic activity, i.e., the *governance structure*, should be devised. In other words, given the rules of the game, how the game is played is studied (Williamson 1998). The methodological foundation of this structural analysis is transaction cost economics.

The final level of analysis addresses the *allocation of resources* and their employment. The building block of this analysis is neoclassical economic theory and, as far as incomplete information is concerned, agency theory. The relevant parameters are relative (market) prices and associated quantities, as well as effective incentive alignments (Williamson 1998).

Since the markets for genetic resources essentially rely upon the definition of some form of exclusive property, my the analysis continues in Section 3.1.2 by describing the current international property right regime on genetic resources and genetic information. In doing so, I introduce simultaneously the prevailing governance structure on transactions with genetic resources. In Section 3.2.1, I define a partial equilibrium framework in order to analyze the impact of market transactions with genetic resources on biodiversity conservation in an idealized fashion. In Section 3.2.2, I briefly delineate conceptual questions on transaction costs and organizational structures in the context of the trade in genetic resources, before Section 3.3 continues with an empirical description of the market.

3.1.2 Property Right Regimes

Against the background of neoclassical theory, nonrivalry with respect to genetic information and exclusiveness with respect to both the material and information raise normative questions regarding the efficient allocation of both biological resources and resources in biotechnological R&D.

In an efficient allocation, the private costs of resource use should be equal to the social costs, or equivalently the social value is internalized in the private resource management. Since genetic resources are rival in their use and can be made excludable, the definition and enforcement of exclusive *private property rights* for the material seems appropriate. However, since genetic resources display a certain value beyond the resource-use value, an identity between private and social values requires additional policy intervention.

Since genetic information is a driver of the social value of genetic resources, the resource owner could be vested with the exclusive rights to the *embodied information*. However, since genetic information is nonrival in its use, i.e., the social costs of using the information are virtually zero, access restriction and the supply of access for return apparently contradicts efficiency conditions.

In contrast, in the absence of exclusive rights to the information, the private resource owner only incompletely appropriates the values that his resources provide. This imperfection usually diminishes the *private incentive to invest* in the resource's preservation: private investments are consequently below socially optimal levels, with negative impacts upon both the resource stock and the (future) availability of information for R&D (Sedjo and Simpson 1995). This finding applies particularly to unimproved information in wild genetic resources. Considering improved genetic information, incentive problems occur when access to the information is unrestricted, since this makes it impossible for researchers to appropriate the value added of their efforts.

In order to reconcile these core issues or to at least extenuate the conflicts between them, a regime of suitable and well-defined *property rights* has to be implemented and enforced (Siebert 2005: 97–99).

The Property Rights Approach

Property rights to an economic good are typically represented by a *bundle of rights*, namely (1) the right to use the good ("usus"), (2) the right to reap the benefits generated by its use ("usus fructus"), (3) the right to change its form and substance ("abusus"), and the right to transfer part or all of these rights to others (Furubotn and Pejovich 1972, 1974).

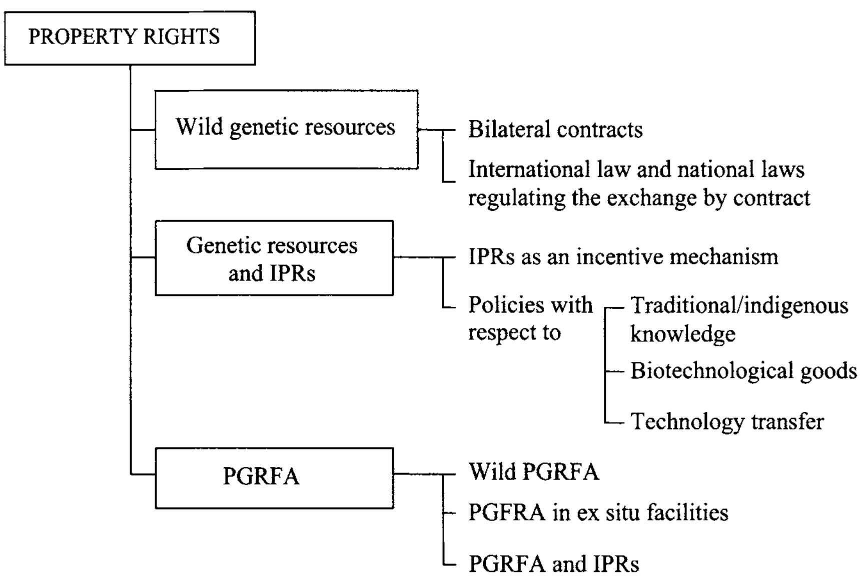
The specification of a property right regime results from existing institutions and social norms (North 1990). The rights may either be *codified by law* or institutionalized by *other mechanisms* (Siebert 2005: 97).

Depending upon the specifications, property rights for an economic good can be distinguished as different types according to access rules (flow management) and conservation rules (stock management). Alternatively, property rights can be distinguished according to specifications regarding the *group size of claimants* and the extent of *use/extraction rights*. Based upon this, a typical classification comprises (1) private property, (2) state property, (3) common property, and (4) open access resources (Heltberg 2002; Siebert 2005: 97f.; Stevenson 1991: 57ff.).

Regarding these categories from a normative perspective, the theory of property rights suggests that rights should be specified in such a way as to achieve the *efficient provision/utilization* of the good considered (Alchian 1965; Demsetz 1967).

Turning to genetic resources and genetic information, property rights differ according to the different areas of (commercial) use, as well as according to the type of information contained in the material. To summarize the property right regime, I consider existing property right regimes for the (1) *material* and (2) *information*,[50] one after the other. Furthermore, since the regime for *PGRFA* shows sector-specific peculiarities that are rooted in its specific economic properties, I address them in an extra description. Figure 6 depicts the structure of the following descriptive analysis.

Figure 6:
Property Rights Regimes with Respect to Genetic Resources



50 Alternatively, information and material can be classified as phenotypes, i.e., individual plants, animals, or other organisms, versus genotypes, i.e., information contained in the genetic constitutions of the species (Sedjo 1992).

3.1.2.1 Property Rights for Wild Genetic Resources

First I consider the common case where property rights apply to genetic resources in in situ conditions, i.e., materials that typically reside on land not used for intensive agriculture or forestry. These materials are often referred to as wild genetic resources (Sedjo and Simpson 1995; OECD 1999).

Property rights for in situ material are either closely connected to the rights to the land upon which it resides or defined more specifically with respect to the resource itself (Janssen 1999). Such specific resource rights can, in principle, either be assigned to the landowner or anybody else,[51] i.e., individual rights or subsets of rights can be allocated to the different stakeholders who manage and/or use the available resource stock and its embodied genetic diversity.

Generally, the literature discusses three alternative types of property right regimes for genetic resources (Sedjo and Simpson 1995; Lerch 1996):

- There are *no restrictions* for the access to and use of the resources, as well as no regulations to conserve them. This is considered to be equivalent to the case where property rights are absent.
- More or less exclusive property rights are defined. Bilateral *contracts* are concluded in order to trade access and use rights to genetic resources.
- A *new property right* for genetic resources is defined that specifically addresses the information embodied.

Unrestricted access and use rights and the lack of an arrangement for conservation imply that genetic resources are subject to an open access regime. For the general case of biological resources, it is shown that an open access right regime leads to overexploitation and resource depletion (Hardin 1968; Heltberg 2002).

The threat of resource depletion combined with the irreversible loss of genetic information on the one hand stems from the uncontrolled extraction of the genetic material itself. On the other hand, depletion is attributed to the conversion of in situ habitats into cultivated land, which is usually associated with the displacement and elimination of genetic diversity among living organisms. This latter aspect is relevant in practice: conserving wild genetic resources generates opportunity costs, since the in situ habitats can also be converted and employed for alternative human use (Sedjo and Simpson 1995). Since it is likely that the conservation costs, incurred by private landowners are smaller than the social costs due to a loss of genetic diversity (and of biodiversity in general), an open access regime for genetic resources cannot provide for their efficient provision and utilization.

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51 Generally, resource users can be described according to the number and types of rights they possess. Their position can be classified as owner, proprietor, claimant, or authorized user (Schlager and Ostrom 1992).

Considering the remaining property right regimes, the literature discusses whether they can provide the *efficient allocation* of wild genetic resources, as well as of natural areas in alternative use. The task, in this regard, is to allow the holders of property rights to natural areas to appropriate (a proportion of) the value these genetic resources provide so that right holders have an incentive to invest in the preservation of genetic diversity and its in situ habitat (Swanson and Göschl 2000). In other words, efficiency considerations with respect to resource management interact with the *distributional issue* of how to share the economic rent associated with (naturally occurring) genetic information and/or the products developed from it (Sedjo 1992; OECD 1999).

Since, in practice, the definition of a property right regime that addresses wild genetic information faces strong restrictions imposed by the existing international regime of intellectual property rights (IPRs), the public discussion focuses upon the design of bilateral contracts to trade property rights to genetic resources.

Bilateral Contracts on Wild Genetic Resources

Bilateral contracts on genetic resources follow the mechanism suggested by *Coase* (1960): when exclusive property rights are specified and enforceable, any inefficiency due to the existence of externalities can be corrected through negotiations between the stakeholders—provided that transaction costs are not prohibitive (see Section 3.2.2). Negative externalities to users of genetic resources are represented by the loss of genetic diversity through overexploitation and land use changes. When property rights are properly specified and access restrictions to genetic resources or their natural habitat are in place, the parties involved negotiate the type of land use. Depending upon how the property rights are allocated, *negotiations* either arrange for conservation of natural areas or for the permitted depletion of the biological resources hosted by the areas.

Considering *bilateral contracts* in practice, property rights are typically assigned to private landowners (private property) or governmental authorities (state properties). Private landowners in particular offer rights to access a natural area or collect and supply genetic resources in return for various forms of compensation. The buyers of the rights/resources raise the amount of compensation through the sale of products developed from the genetic material and the information contained. For a profitable exchange, this compensation must exceed or at least be equal to the net revenues the landowner can obtain from alternative use.

In this understanding, in situ providers of genetic resources play the role of "gatekeeper," i.e., they offer access/collection rights to unimproved and unscreened genetic raw materials. The compensation represents the price for the right to access. In this context, it is sometimes suggested that the providers may undertake additional activities in the multistage R&D processes with genetic resources in order to increase their revenues. By *vertically integrating* early stages of R&D within the individual domain, an in situ provider could receive a higher price for (processed) genetic material and, thus, would be able to appropriate a larger share of the economic rent of genetic information (Artuso 2002; Dutfield 2000a).

Vertical integration in this respect is considered a substitute for bilateral contracts on collection rights in the initial stage of the R&D process (Simpson and Sedjo 1994). If the in situ provider integrates stages of production that are nearer to the market for biotechnological final products, this is named *upstream integrating*. Alternatively, the producers in the final stages of the industry may merge downwards in order to control the sources of material supply (*downstream integrating*) (Swanson and Goeschl 2000).

Property Rights for Genetic Resources in International Law: The Role of the CBD

On a conceptual basis, negotiations on access and use rights for genetic resources do not demand a (new) property right regime that is, for instance, codified by law. It suffices if the allocation of de facto property rights is mutually accepted (Lerch 1994; Sedjo and Simpson 1995). In practice, nevertheless, codified property rights to genetic resources have indeed been defined. Legal frameworks that are implemented on a subnational or national level are derived from regulations on an international level that, in turn, are predominately determined by the provisions of the CBD.

- The CBD represents an *international property right regime* that has resulted from negotiations between the sovereign countries, i.e., it represents a collectively agreed upon regime.
- The CBD assigns *sovereign rights* for the resources on national territory to the governments.[52] More precisely, the CBD assigns "the authority to determine access to genetic resources" to the national governments (CBD Art. 15(1,3)). In doing so, the CBD proceeds on the assumption that property rights are primarily national governmental rights. Furthermore, it rejects a common property approach as it is represented by the concept of the common heritage of mankind (Lerch 1998; Wolfrum et al. 2001: 47ff.).

52 More precisely, these are those resources for which the country is the country of origin or that are acquired in accordance with the access provision of the CBD (CBD Art. 15(3)).

- The CBD addresses the international and therefore cross-border utilization of genetic resources. In order to enable an international exchange, countries have an *obligation to facilitate access* to genetic resources (CBD Art. 15(2)). In this respect, the CBD balances the interests of the individual resource country for sovereign rights and the collective interests of countries for access to the world's gene pool.

Further provisions specify the scope of the property right regime:

- Access should only be granted for "*environmentally sound uses*" (CBD Art. 15(2)).
- The provisions only refer to the exchange of genetic resources *after the signing of the CBD*. Genetic materials that users have acquired beforehand and that are currently stored in ex situ collections are not subject to the CBD regime. This applies especially to ex situ PGRFA for plant breeding activities. Although these resources are subject to the CBD, they are addressed in other multilaterally agreed regimes. In addition, recent policy efforts have aimed to attain consistency among the provisions in the different regimes (see Section 3.1.2.3).

While it is often argued that the CBD redefines the property rights to genetic resources in that it replaces a previously (de facto) open access regime for such resources (e.g., Dhillion et al. 2002; Artuso 2002), Wolfrum et al. (2001: 82f.) highlight the connection between biological and genetic resources and remark that the property rights to biological resources have already been assigned to the national states. In this regard, the CBD rather reaffirms the existing sovereign rights.

The difference to the pre-CBD situation apparently lies in the multilateral regulation of the transfer of rights. The CBD defines a new legal framework that has to be considered when concluding bilateral contracts between governmental and/or private organizations. The building blocks of this framework are the principles of *mutually agreed terms* (MAT) and *prior informed consent* (PIC) (CBD Art. 15(3,4)). Countries that are parties to the CBD are called upon to establish an appropriate legal framework that regulates access to their genetic resources in such way that these principles are satisfied in every bilateral contract (OECD 1999, 2003b; Laird and Wynberg 2003).

Although the CBD provisions do not further describe how to implement the principles in practice, it is commonly agreed that the provisions effectively stipulate that the exchange of genetic material has to follow a somehow co-operative model between the provider and user of genetic resources: individual holders of property rights for the resources should be fully aware of the economic circumstances when exercising their rights; it should not be possible to deceive them as to the true value of their resources (Glowka 1998; Wolfrum et al. 2001: 84f.).

In effect, the principles of MAT and PIC represent instruments to achieve fairness and equity in the sharing of the benefits that arise from the utilization of genetic resources. Consequently, with reference to benefit sharing, the corresponding regulations directly address one of the CBD's three core objectives (CBD Art. 1(1)) and are thus central to global biodiversity policy.

In this context, it is argued that the obligation to cooperate that is incorporated in the CBD goes beyond one-time monetary compensation for the provision of genetic materials. Moreover, the legal framework implies a close connection between the provision of genetic resources on the one hand and the entitlement of providers to participate in the knowledge and technologies derived from the resources on the other (OECD 2003b; Wolfrum et al. 2001: 84f.).

The call for cooperation has to be considered against the background that providers are usually located in biodiversity-abundant developing countries, while commercial users are primarily located in the developed countries, and that, in this respect, the provider countries' interest in economic development should be taken into account. Accordingly, benefit sharing arrangements in the context of the exchange of genetic resources should include elements that address the access to and transfer of technology, the exchange of information, and/or cooperation in a technical and scientific regard (Dutfield 2000a).

Property Rights in National Law: Provisions on Access and Benefit Sharing

Given the provisions of the CBD on defining access and use rights and benefit sharing regulations, national governments are called upon to translate the international terms of reference into national laws and practical domestic policy.

Since the CBD assigns the property rights to the governmental level, each provider country has to decide whether to retain rights within the public sector or to assign them to communal or private landowners (OECD 2003b). Additional regulations on land use may be implemented in order to ensure the conservation of genetic resources. Finally, political processes on a local or national level have to be initiated in order to define who owns the fundamental rights to genetic resources, i.e., who are the local stakeholders in a specific case and what kind of veto right is granted to each of them, who has to be informed, and whose permission is required when accessing the resources (Glowka 1998; Laird and Wynberg 2003).

Since the CBD provisions on access to genetic resources and benefit sharing (ABS), particularly on the subject of benefit sharing, as well as on the PIC and MAT procedures, are somewhat broadly defined, there have been considerable debates upon these topics. As a reaction, there have been efforts in the forum of the CBD to provide information in order to develop a unique framework for national legislation. These efforts were finalized in 2002 in the Bonn Guidelines on Access to Genetic Resources and Fair and Equitable Benefit Sharing (Linarelli 2004).[53] The guidelines are generally nonmandatory in character and aim at the minimization of transaction costs that are associated with the use and exchange of genetic resources, and they particularly support the provider countries in specifying national laws. In addition, the guidelines highlight the responsibility of the user countries and suggest implementing mechanisms that verify the compliance of commercial users with the CBD provisions on ABS and the provider countries' legal instruments.

In spite of these guidelines, there are in practice still differences in the regulations on the access to and use of genetic resources among the provider countries. In order to coordinate their interests in combating the unauthorized acquisition of genetic resources and enforcing an international regime that enables the inequitable sharing of benefits, 17 provider countries with exceptionally large biodiversity endowments have formed the group the *Like-Minded Megadiverse Countries* (LMMC) (Stevenson 2002; see Section 2.1.2).

In order to translate the international provisions, the provider countries either modify existing national regulations or adopt new ones. A database by the Secretariat of the CBD (2005) summarizes *evidence on legal provisions* that deal with the ABS issue and that were implemented after the signing of the CBD in 1992. I have calculated the number of new regulations in CBD signatory countries according to the year of their coming into force. Figure 7 indicates the results.

It shows that about 30 regional, national, and subnational level regulations have been introduced since 1992. More countries have already adopted new regulations that, however, have not yet come into force. Concerning the *regional distribution of new regulations*, it turns out that more than one half of the regulations have been implemented by Latin American countries (15). Some regulations are observed for African countries (5), and countries in Asia and Oceania (5) (Secretariat of the CBD 2005). In total, the establishment of national ABS regulations worldwide is not yet complete, but rather still in flux; there is evidence that many provider countries have not yet developed an ABS regime in accordance with the CBD.

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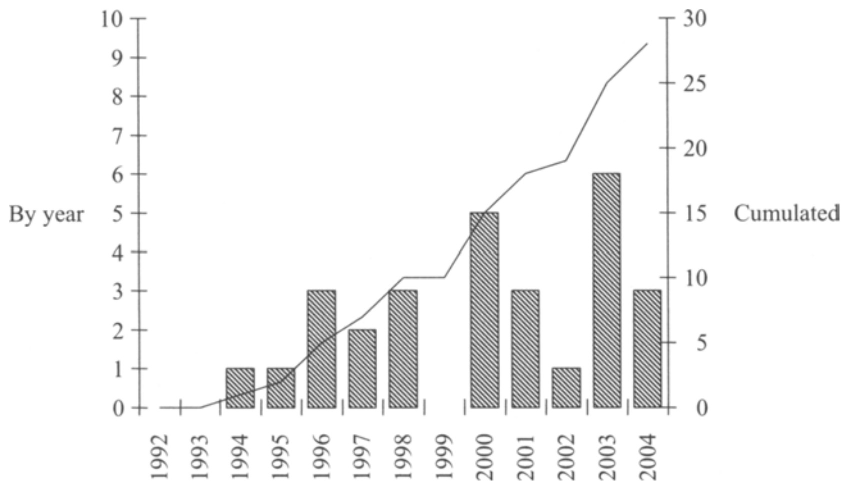
53 These guidelines also represent a kind of harmonization of national access and benefit regulations and, in this regard, address fears that regulatory competition between supplier countries could eventually lead to lax national provisions that in the end would inhibit fairness and endanger the commonly agreed upon objectives of the CBD (ten Kate 2002).

Figure 7:
Regional, National, and Subnational Regulations on Access and Benefit Sharing



Notes: Columns describe regulations by year of entry into force. The line indicates the cumulated number of regulations.

Source: Secretariat of the CBD (2005), own representation.

Several comparative studies in economics and other social sciences analyze specific national regulations, as well as contracts concluded upon the basis of these regulations. The empirical investigation of trade in genetic resources in Section 3.3.2 draws partly upon this literature. However, a detailed description of this literature lies outside the scope of this study.[54]

3.1.2.2 Intellectual Property Rights and Genetic Resources

Since the value of a genetic resource is essentially determined by its informational content, property rights that address information as an intangible good, i.e., intellectual property rights (IPRs), have an impact upon the allocation of

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[54] Further information is provided in Glowka et al. (1997), Liebig et al. (2002), Ramanna and Smale (2004), Reid et al. (1992), Richerzhagen and Holm-Mueller (2005), and Richerzhagen (2003), to mention some studies. A synthesis of national legislation is provided in UNEP/CBD (2005). SIPA (1999) and UNEP/CBD (1999) review bilateral arrangements that have been reached against the background of these provisions.

genetic resources. Before considering IPRs in the context of biodiversity, the general aim and function of IPRs as presented in the literature is briefly reviewed.

The IPR System as an Incentive Mechanism

A property right regime for information goods aims at creating incentives for the efficient provision and utilization of information (Swanson and Goeschl 2000). Problems arise from the fact that information is nonrival in its use and *exclusion* from its use can be *prohibitively costly* for the inventor/provider.

The inventor has to cover his costs from revenues he receives for a tangible good that owes its quality to the invented information. Once the good is outside the inventor's domain, anyone else can *reveal* the information, *reproduce* it at practically zero cost, and *compete* with the inventor in the market for that information good. As a consequence, the inventor may obtain insufficient returns. In anticipating this outcome, he underinvests in research or completely withdraws, the result being that the invention remains incomplete and the potential welfare improvements for society are not realized.

To rule out such a suboptimal outcome, a regime of IPRs can be established that enables the inventor to control the subsequent use and marketing of the information. This holds although the information is in principle accessible once the information good is introduced in the market. Such an exclusive marketing right provides the inventor with an opportunity to earn reasonable returns in the market. The granting of such a specific right effectively functions as a mechanism to create *incentives for investment* in the search for new information (Varian 1999; Besen and Raskind 1991; Arrow 1962).

Imperfections can arise in two respects. First, the actual incentives for the researcher strongly depend upon the extent to which the exclusive rights are actually *recognized* by the competitors on the market and monitored and *enforced* by the government. If, for example, competitors are able to de facto acquire and use the information on an illegal basis, the inventor's returns are reduced, with the consequence that the ex ante incentive to research is weakened (Varian 1999).

Second, if the inventor is generously vested with an exclusive right to use specific information and this right expands across many related uses and/or is extended over a long period of time, *subsequent innovations*, which are desirable from a societal perspective, can be *impeded*. In this regard, the incentive to research and create new information generally has to be balanced with the deadweight losses that result from the inventor's exclusive rights (Rausser and Small 1996). These aspects are addressed in debates on the optimal length and scope of IPRs (Jansen 1999; Chou and Shy 1993; Klemper 1990).

In practice, several *forms of IPRs* exist. The familiar examples are (1) patent rights, (2) copyrights, (3) trademarks, and (4) trade secrets (Besen and Raskind 1991). All forms can be part of an IPR regime simultaneously, i.e., the forms are not mutually exclusive in national or international law. The different forms of IPRs have in common the fact that they define certain exclusive use rights to the information good, whereas the good itself does not need to be in the domain of the right holder (anymore) (Swanson and Göschl 2000). Differences between the forms are represented by differences in the extent of the right holder's control over use by other people. A patent right permits the most extensive control in this respect.

IPRs and Genetic Resources: Key Issues

Several information goods are subject to IPR systems. Examples are books, movies, and music, but also software, machines, and pharmaceuticals. In the biological realm, IPRs have an impact upon the allocation of several goods, including genetic material, as well as goods that are related to it.

The complex system of rights that has developed has, in this respect, been subjected to many analyses in several disciplines of the social sciences (Dedeurwaerdere 2005; Dutfield 2002; Görg and Brand 2000). An in-depth review of this literature, however, lies outside the scope of my study. Instead, I delineate the major topics by distinguishing four key issues.

The first and second issues relate to the granting of IPRs for goods that are closely connected to biodiversity.

- First, an IPR can, in principle, be defined for the *material that embodies the genetic information*. The information embodied can, in this regard, be further classified as wild or naturally occurring information and information created through human interference (invention). Controversies, in this respect, arise as to where to draw the line between the two concepts, i.e., how to assess the importance of man in the creation of the information that serves as the basis for justifying exclusive rights.
- Second, an IPR can also be granted for the *information on the use of genetic material*, i.e., the information on the function of specific genetic material that, for example, enables the researchers to ex ante distinguish it from other less promising material or successfully modify the material. Traditional and/or indigenous knowledge refers to this type of information. Since this knowledge has already been created and is sometimes accessible for free, it is discussed as to whether an IPR-like system should be established that enables exclusive rights to control the subsequent use of the knowledge. This also raises the question as to whether to ex post acknowledge efforts in the creation and recent maintenance of information.

The third and fourth issues relate to the impact that IPRs to other information goods have upon the management of genetic resources.

- This is, first, the impact of *IPRs to biotechnological products* upon (1) the use of genetic resources as an input into R&D, as well as the (2) use of domestic genetic resources as inputs into local (agricultural) production. On the one hand, several studies suggest that a strengthening of such IPRs increases the demand for genetic resources and, hence, raises the resource price; on the other hand, IPRs may reinforce incentives on a local level to replace domestic genetic diversity with internationally traded but genetically less diverse resources.
- Furthermore, *IPRs for other man-made goods*, such as machines or scientific knowledge, influence the preferential access to technologies and *technology transfer* that is provided for in access and benefit sharing arrangements and that, for this reason, indirectly influences the incentive to preserve genetic resources.

In the following, I briefly summarize the scientific and political discussion on each of these four issues and describe the relevant provisions in international law. The major focus in this context falls upon

- whether sufficient incentives are created in order to guarantee the efficient provision of the goods involved, particularly genetic resources and, connected with this,
- whether the distribution of the economic rent of genetic information among the stakeholders is considered fair and equitable.

An IPR to Genetic Resources?

So far, I have assumed that the resource owners conclude bilateral contracts that provide them with returns for the sale of the right to access their resources. Alternatively, it is conceivable that property rights are arranged in such a way as to vest a right holder with the right to control the subsequent use of the genetic information embodied in his resources. When such rights are assigned to the landowners upon whose land the resources reside, they may capture the rent associated with genetic information more effectively, as compared to arrangements based upon bilateral contracts. Accordingly, more substantial incentives for private conservation may be created (Sedjo and Simpson 1995; Subramanian 1992).

However, the establishment of an IPR-like regime for wild genetic resources faces strong practical barriers: there are difficulties in defining the rights to genetic material unambiguously, which is nevertheless necessary to support an efficient allocation effectively. Ambiguity results from the fact that in situ material of an identical genetic constitution appears at several places (and, therefore, in the domain of many potential right holders). Consequently, it is hardly feasible to assign exclusive rights for the information embodied (Sedjo 1988).[55] As a way out, exclusive rights may be assigned to groups or communities but such an approach may be hampered by practical problems in monitoring and enforcing the rights. In addition, a mechanism has to be implemented that allocates the payments to the right holders and, therefore, creates effective incentives for conservation on a local level. Such a mechanism is likely to be constrained by large transaction costs (Subramanian 1992).

In addition to these problems of feasibility, a general caveat to this type of property right is made on legal grounds. To obtain intellectual property protection, a researcher typically has to satisfy certain requirements. For patent protection, for example, the researcher is asked whether the information in question indeed represents a *true invention* or whether it is more of a discovery (Walden 1995). Existing IPR systems are based upon the premise that only innovative efforts should be acknowledged in the granting of patent rights. This applies primarily to engineered, modified, or improved genetic material but not to unmodified wild genetic resources (Sedjo and Simpson 1995).

This point of view is also reflected in societal reservations towards an exclusive expropriation of the values of natural resources, which represent a creation by nature as opposed to man. These reservations are not actually confined to natural material but are also influenced by the controversial discussion as to when IPRs for invented/modified materials are generally justified (Dunleavy and Vinnola 2000; Gadgil 1996).

Since information goods are subject to cross-border trade, countries have agreed upon an international regime of IPRs that is represented by the multilateral *Agreement on Trade-Related Aspects of Intellectual Property Rights (TRIPS)*. This agreement, which is binding for all members of the World Trade Organization (WTO), aims to establish the worldwide protection of intellectual property. The member countries are called upon to implement national laws on IPR that are consistent with the standards defined in TRIPS (Stegemann 2000).[56]

TRIPS includes several provisions relevant to the issue of "life patenting" (TRIPS Art. 27). Genetic materials are distinguished according to their bio-

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[55] In addition, since many genetic materials have not yet been discovered or at least not described, it is difficult to define property rights to goods whose quality is not yet known (Sedjo 1988).

[56] International arrangements for IPRs can look back over a long history. The starting point was the 1883 Paris Convention for the Protection of Industrial Property. The World Intellectual Property Organization (WIPO) serves as the institution that administers the relevant international agreements.

logical taxa. Based upon this, different obligations and options to make genetic materials subject to patent protection are specified. National laws must, in principle, include provisions on the patent protection of biotechnological goods and processes (TRIPS Art. 27(1)). However, plants and animals can, per se, be excluded from patentability. The same applies to traditional breeding processes for the production of plants or animals, as well as for "diagnostic, therapeutic, and surgical methods for the treatment of humans or animals" (TRIPS Art. 27(3a), (3b)).

This exemption does not include microorganisms for which patent protection has to be implemented. The same applies to microbiological processes that may relate to the production of plants, animals, or other organic material (Wolfrum et al. 2001: 68ff.; Bhat 1996). Furthermore, regarding plant varieties as entities of PGRFA, TRIPS's provisions either allow patent protection or establish a sui generis IPR system, i.e., a regime "of its own kind," or a combination of the two forms (TRIPS Art. 27 (3b)).

On a national level, IPRs for derived lead compounds, processes, or final products may be assigned to intermediaries, such as screening agencies or private and public R&D institutions. Although the TRIPS agreement aims at the harmonization of national IPR systems, there are still differences between countries for cultural or historical reasons. Previously, many developing countries did not recognize IPRs as they are defined in the developed countries (Bhat 1996).[57]

When examining the impact of IPRs upon the conservation of (in situ) genetic resources, it is shown that the current IPR regime supports the capture of economic rents from genetic information more through the outputs in the final stages of biotechnological production. Incentives to provide and maintain naturally occurring genetic information in the initial stages are not addressed. Consequently, the current IPR regime does not seem to support biodiversity conservation in this respect (Sedjo 1992; Swanson and Goeschl 2000). Furthermore, when considering the political level, it is argued that the recent international regulation of IPRs effectively led to the expansion of the developed countries' property rights to developing countries (e.g., Koopman 2005). This is of importance to the distribution of the rent of genetic information among the stakeholders.

IPRs and Knowledge of Biodiversity and Genetic Resources

Information is embodied in the genetic material but information also relates to its use. In order to make this information become instrumental, it is transformed

[57] The implementation of national IPR systems with regard to biotechnological goods has been described and analyzed in several studies (Dunleavy and Vinnola 2000; OECD 1996, 2002).

into knowledge. Regarding in situ genetic resources, their management frequently depends upon the knowledge that has already been created by *indigenous communities*, which traditionally use the resources and often live in or around the resources' habitats. This knowledge also enhances the value of specific genetic material for R&D purposes and, thereby, indirectly influences the incentives to preserve and offer access rights (Sheldon and Balick 1995).

The stock of indigenous knowledge typically is not static, but, rather, includes ongoing innovations (Downes and Laird 1999). It is then evident that control over this dynamic knowledge is connected to the incentive to preserve the genetic resources the knowledge refers to and the incentive to maintain the knowledge itself (Downes 2002). Access to indigenous knowledge (and to other traditional knowledge) is not usually restricted, i.e., the knowledge either belongs to the public domain or is subject to a specific cultural ownership regime that emphasizes the collective use of the knowledge and the responsibility to maintain it (Dutfield 2000b; Downes 2002).[58]

The debate on indigenous knowledge, as well as traditional knowledge in general, is dominated by considerations on equity and fairness. There are cases where technology-intensive R&D firms have developed biotechnological goods for which they have been awarded exclusive patent protection—even though the inventive step in the product development has been derived (in part) from indigenous/traditional knowledge. In this respect, *indigenous rights to knowledge* have been ignored. Even worse, the patent right granted to an R&D organization may allow the restriction of the communal use of the biological material related to the knowledge (Koopman 2005; Bhat 1999).

Given these conflicts, the legal and political discussion has focused upon (1) how indigenous communities can protect their knowledge by means of an internationally acknowledged IPR system and (2) whether certain patents on life forms indeed satisfy the basic requirements for international patent protection. These discussions frequently take place against the background of different cultural attitudes between the indigenous people and local communities on the one hand and modern societies on the other (Byström et al. 1999). It is argued that the efforts towards establishing property rights that are acknowledged worldwide will finally impose "one conception of ownership and innovation on a culturally diverse reality," with the consequence that the private sector that makes use of goods under an IPR regime will benefit the most (UNDP 1999: 70).

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[58] Indigenous knowledge is considered to be knowledge that somehow is assigned to a specific group or community considered "indigenous." Traditional knowledge, in turn, refers to the creation of knowledge in the past and its transfer from generation to generation. (Wolfrum et al. 2001: 42f.) In this respect, indigenous knowledge can represent traditional knowledge simultaneously, i.e., it represents a subset of traditional knowledge (Mugabe 1998).

Regarding patent protection, a crucial criterion is whether the invention is genuinely new (Walden 1995). While it is assumed that indigenous knowledge typically lacks *novelty*, R&D firms that own a patent right argue that they have indeed added an inventive step and that therefore granting patent protection to them is justified. Nevertheless, in the past, indigenous communities have successfully challenged controversial patents (Downes 2002; Prakash 2000).[59]

Because patent protection is, per se, not possible for indigenous knowledge, *alternative models of an IPR system*, such as trademarks or a sui generis ("stand on its own") system, have been discussed. Additionally, it has been considered whether to implement intellectual property protection for indigenous knowledge as part of (digital) registers or libraries. Such instruments intend to enable indigenous communities to control the use of their knowledge and, if so desired, participate adequately in the returns from commercialization. The legal and economic viability of such an approach is still the subject of debate (Downes and Laird 1999; Koopman 2005).

On an international level, efforts have been made to raise the issue of indigenous communities and their fair participation in the sharing of benefits arising from the utilization of their traditional knowledge. With regard to IPRs, the discussion has taken place in the forum of the WTO (with respect to the TRIPS arrangement), the World Intellectual Property Organization (WIPO) and the United National Food and Agriculture Organization (FAO).

In addition, the concerns of indigenous communities are also supported in the CBD, without directly addressing the IPR issue. The CBD assumes a close connection between biodiversity conservation and the maintenance of indigenous knowledge. According to CBD Art. 8(j), each signatory country has an obligation to "respect, preserve and maintain the knowledge, innovations and practices of indigenous and local communities," however, only "as far as possible and as appropriate." In order to fulfill this obligation, the countries may use specific IPRs as a tool to strengthen the position of indigenous peoples.

To resume, the literature suggests that the role of IPRs in the context of indigenous knowledge is ambivalent. On the one hand, the worldwide expansion of exclusive IPRs seems to pose a threat to the interests of indigenous communities and, thus, indirectly affects the traditional use of biodiversity, as well as its conservation; on the other hand, the intellectual property institution could also serve as an instrument to enhance the ability of local communities to enforce their cultural and economic interests (and, if so desired, capture the economic rent associated with their knowledge). Potential conflicts with biodiversity conservation could be mitigated in this regard.

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59 Well-known examples are the turmeric patent, the ayahusca patent, or patents related to the neem tree (Downes 2002; Prakash 2000).

The Impact of IPRs to Biotechnological Goods

In spite of the conflicts in granting IPRs in the context of biodiversity, patent protection exists for many goods produced using biological input and/or biotechnological applications. As argued above, intellectual property protection supports the incentive to invest in R&D.[60] If IPRs, in this respect, lead to an increase in R&D activities, this also increases the demand for the inputs used in the research process. For biotechnological R&D, this includes the demand for biological materials (Droege and Soete 2001; Sedjo and Simpson 1995).

An *increasing demand for genetic resources* increases their relative scarcity and therefore the suppliers of genetic resources may gain bargaining power. If they manage to obtain higher resource prices on the market, the preservation of natural habitats as bioprospecting areas can become the most profitable land use. In this respect, the enforcement of IPRs would indirectly assist biodiversity conservation. Furthermore, suppliers of wild genetic materials may receive revenue-dependent payments or even claim a joint ownership of the IPR for the resulting final product. When intellectual property protection leads to increasing revenues in the output market, the suppliers of genetic resources directly benefit from the enforcement of IPRs.

However, whether the impact of reinforced IPRs upon the demand for genetic material is indeed substantial needs to be examined. Moreover, the increase in the demand for R&D input induced by IPRs may concentrate more on other input, such as skilled labor and machines, than on biological resources (Swanson and Goeschl 2000).

IPRs for biotechnological goods can influence biodiversity conservation in an additional way: the stringent enforcement of IPRs stimulates the development and supply of new biotechnological goods. These goods often *substitute conventional biotechnological goods* that represent carriers of unique genetic diversity, such as traditional crop genetic resources or plants of botanical medicine. The replacement of these goods within biodiversity-abundant developing countries occurs because these countries start to establish their own industries for the development of new but genetically less diverse biotechnological goods. Furthermore, the worldwide recognition of IPRs promotes the international trade in the newly developed goods. Within domestic markets, the domestically produced conventional goods increasingly compete with those that are traded internationally and whose production relies on capital-intensive technology.

To assess the impact upon the conservation of biodiversity in general, it is crucial how the domestic conventional goods are related to conservation. If there

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60 Empirical evidence for expanded private R&D activities can be observed for the area of plant breeding and for research with microorganisms (Swanson and Goeschl 2000; Aylward 1995).

is a close link and the foreign goods dominate the domestic ones, the enforcement of IPRs has an adverse impact upon conservation: the supply of new substitutes/imported goods reduces the value of the local biological resources employed in domestic production, with the consequence that resource owners have an incentive to abandon the management of these resources or deplete the resource stocks (Bhat 1996, 1999).[61]

While the ambiguous impact of IPRs for biotechnological goods upon the incentive to conserve and use genetic resources/biodiversity suggests that these intellectual property institutions should be designed carefully, it has to be acknowledged that IPRs serve other policy objectives in the first place. National governments, which are responsible for the design and enforcement of the rights, for example, aim at establishing a biotechnological sector in order to foster economic development. If the recognition and enforcement of IPRs is a helpful tool in this regard, policymakers may, to some extent, tolerate conflicts with biodiversity conservation (Bhat 1996, 1999).

Finally, in the international fora, it is discussed whether IPRs for biotechnological goods can be used as an instrument to assist compliance with the benefit sharing provisions of the CBD. More specifically, one proposal is to make the *granting of patent protection* dependent upon whether the researcher who has filed a patent application can verify that the genetic resources used in the R&D process have been acquired in *accordance with the CBD*'s provisions on ABS and national provisions in the country where the resources originated (Wolfrum et al. 2001: 105; Downes 2002).

IPRs, Access to Technology, and Biodiversity

Biodiversity conservation and IPRs are also connected to each other in the context of the *technology transfer* that is provided for in access and benefit-sharing arrangements. The CBD assumes that developed countries receive (wild) genetic resources and provide patent-protected technologies to developing countries that are rich in biodiversity but lacking in technology. The use of these technologies (inter alia) supports the conservation of the environment (Art. 15 (7) CBD).

In a narrow definition, technologies transferred according to the principles of access and benefit sharing can refer to technologies that aim at the conservation and sustainable use of biodiversity. In a broader definition, the transfer may

61 A typical example is the use of traditional plant varieties, which are increasingly being replaced by new plant varieties. As a consequence, unique genetic information is irreversibly lost and evolutionary processes in in situ habitats are disturbed or interrupted. The extent to which the replacement is indeed attributable to IPRs needs to be investigated, as does the extent to which replacement is responsible for the loss of genetic information (Zilberman et al. 2004; Dutfield 1999).

include technologies that primarily aim at assisting developing countries in becoming integrated in the global economy but that, at least, do not cause environmental damage (UNEP/CBD 1996b). Examples are technologies related to the management of biodiversity, especially in protected areas, or any industrial technology that is owned by the commercial users of genetic information and that is of interest to the suppliers of (in situ) genetic resources.

An exchange of technology and genetic material effectively only occurs if it is profitable for both sides. For the owner of technology, profitability crucially depends upon whether the IPR for the technology is recognized in the recipient country (WTO/CTE 1995). If this is not the case, there is a danger that the technology will be imitated and supplied on the market below the current price. In this respect, the owner may refuse to hand over the technology, with the consequence that the well-being of the provider of genetic resources will not be improved either. In contrast, the recognition and enforcement of international IPRs in developing countries may increase the likelihood of a technology transfer to these countries.[62] When the prospect of a technology transfer, in turn, promotes a developing country's incentive to conserve, IPRs indirectly support conservation.

By definition, IPRs for a foreign provider of technology impose restrictions upon use rights in the recipient countries. Depending upon the scope of these restrictions, there are certain objections in these countries. It is feared that due to strong IPRs, e.g., broadly defined patents, a technology transfer could affect *competition on domestic markets*, as IPRs support the market power of the foreign technology provider. Moreover, it is claimed that a foreign IPR holder can use his exclusive rights as a defensive strategy: in order to strengthen his individual market position, the protected technology is either retained or no permission is granted to researchers to improve and develop the technology for domestic demand (Dutfield 1999).

This argument has to be seen against the background of the present, imbalanced distribution of IPRs and, particularly, patents worldwide, since the vast majority of patents are held in the developed countries.[63] In order to enable the poor in the developing world to benefit in particular from newly developed but

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[62] It is difficult to verify the implied relationship between stronger IPR and technology transfer empirically. The empirical literature has previously focused upon the relationship between IPR enforcement and inward flows of foreign direct investment. Although a positive link between the two is not rejected, it is suggested that a complex network of many factors, such as market structure and national policies with regard to market liberalization, determines the extent of the technology transfer (Maskus 2000, 2005).

[63] About 97 percent of all patents belong to residents in the developed countries. Developed countries even hold 80 percent of the patent rights granted in developing countries (Butler 1998; UNDP 1999: 67f.).

IPR-protected technologies, it is often argued that a fair solution between the interests of the inventor/provider of the technology and its users in the developing countries has to be found concerning the recognition of IPRs (Dutfield 1999).

An instrument proposed in this regard is *compulsory licensing*. With such licensing, the exclusive rights of a patent holder are restricted such that use rights are transferred for a price determined by the government, without the consent of the patentholder (Dutfield 1999). Such a policy typically raises objections on the part of the technology inventors. It is argued that compulsory licensing would lead to the improper transfer of their technologies and that it may conflict with the existing trade provisions of the WTO regime (Prakesh 2000). In a modified form of compulsory licensing, a developed country's government may consider subsidizing firms that transfer their technology voluntarily. In this regard, the government takes on a stronger role in the usually decentralized exchange. Technology transfer can be regarded as a specific form of in-kind transfer organized on an intergovernmental level (Stähler 1994).

Considering the impact of transferred technology upon the incentives to conserve and the role of IPRs, two things are of importance: first, the types of technologies that actually promote the conservation and sustainable use of biodiversity have to be identified and the extent to which effective technologies are part of the technology transfer agreements needs to be estimated. Second, since not all technologies are subject to patent protection (anymore), the extent to which the use of technologies relevant for biodiversity conservation is currently restricted by IPRs has to be studied (Dutfield 1999).

According to UNEP/CBD (1996b), the relevant technologies relate to the "establishment and management of protected areas," "scientific research-based activities," and the management of the components of biodiversity in ex situ conditions. In addition, the use of industrial biotechnology is considered important for the conservation and sustainable use of genetic resources. There is also some evidence that a substantial share of the relevant technologies is in the public domain, either because a granted patent protection has expired or patent protection has not been sought (WTO/CTE 1995; FAO 1998: 292). The importance of IPR recognition has to be qualified accordingly.

To resume, the diverse relationships between IPRs, technology transfer, and efforts towards biodiversity conservation are too complex to provide a conclusive answer on the impact of IPRs in this regard. Furthermore, empirical evidence that is needed to assess the qualitative role of IPRs is sparse.

Considering the international regulation of technology transfer in the TRIPS agreement, it is shown that, under certain conditions, the suspension of IPRs is permitted ("Other Use without Authorization of the Right Holder," Art. 31 TRIPS). Furthermore, while biodiversity-related technologies are not directly addressed, the TRIPS agreement provides for the exclusion of patent rights for technologies that have a damaging impact upon the (domestic) environment (Art. 27(2) TRIPS) (WTO/CTE 1995).

3.1.2.3 Property Rights to Plant Genetic Resources for Food and Agriculture

The previous description of the property right regime refers to genetic resources in general. PGRFA, as a subset of genetic resources, are subject to some sector-specific regulations for historic reasons as well as for reasons of their specific economic properties.

- In contrast to pharmaceuticals, often the quality of a newly developed plant variety cannot be attributed to individual genetic information isolated and extracted from a clearly identifiable species. Moreover, the value added by a variety results from the arrangement of information as a novel combination of investigated and unexplored gene sequences (e.g., Dutfield 2000a).
- This, in turn, implies that any success in R&D largely depends upon the availability of and access to a large-scale gene pool. Since this gene pool includes previously investigated and/or modified materials as the carriers of useful information, R&D spillovers on the level of firms are seemingly more pronounced than in other sectors that make use of biotechnologies.
- This interdependency among breeding organizations also applies on a national level: while developing countries are the major providers of genetic diversity in general and the commercial users are primarily located in the developed countries, this does not hold to the same extent for PGRFA. On the one hand, many developing countries are not self-sufficient in genetic diversity for plant breeding and demand access to the world's gene pool; on the other hand, developed countries have accumulated replications of a significant share of the world's PGRFA in domestic, ex situ conditions (Dutfield 2000a; Kloppenburg and Kleinman 1988).

Given that the fundamental function of property rights is to promote investment for the efficient provision of an economic good, a property rights regime for PGRFA has to consider their peculiarities while simultaneously rendering effective incentives for R&D in new crop varieties. Because of these particularities, which impose strong requirements for attaining an efficient provision of PGRFA and because of the importance of plant breeding for food production and security, the public sector takes on a strong role in the management of PGRFA compared to other biotechnological industries (Virchow 1999a).

Property Rights to Wild PGRFA

Categories of goods similar to wild genetic resources (see Section 3.1.2.1) and indigenous knowledge (Section 3.1.2.2) can be found in the PGRFA area: first, there are previously unused wild crop genetic resources, i.e., wild material/wild relatives that are genetically related to the major cultivars. Second, there are crop genetic resources (landraces) that owe their quality to human selection at the farm level, i.e., the improvement of the genetic material is not due to capital-based technology input (FAO 1998: 51ff.). Subsequently, in contrast to indigenous knowledge, the impact of traditional use is already materialized in the tangible good.

Considering the three alternative concepts of a property rights regime for wild genetic resources again, an *open access* regime for crop genetic material in in situ conditions is apparently inappropriate for the reasons mentioned above.

In practice, there is little evidence of an open access resource. It is observed, rather, that in situ habitats for unused wild genetic resources are often integrated in protected areas, i.e., the preservation of crop genetic diversity is provided as a joint product of conservation aimed at biodiversity in general (FAO 1998: 54ff.). Since the public sector governs protected areas in the first place, the governmental authorities also hold the property rights to wild genetic resources (see Chapter 4). Furthermore, wild genetic resources are sometimes held in the private domain. This holds in particular for materials that are used in private or communal farming. In this respect, the landowners typically own the exclusive resource property rights.

Any transfer of the right to access and use of wild genetic materials can be arranged in bilateral *contracts*. This applies to the in situ resources in both the private and public hand. On an international level, the International Code of Conduct for Plant Germplasm Collecting and Transfer, adopted in the framework of the FAO, contains provisions to be implemented in national law in order to guarantee the appropriate specification of such contracts (FAO 1993; Barton and Siebeck 1994).

Finally, a property right regime may be implemented that provides the owner of the land hosting the genetic material with some exclusive rights to the informational content of the material (see Section 3.1.2.2). However, such an IPR-like system faces similar impediments to those discussed for the general case, namely difficulties in assigning property rights unambiguously and in being consistent with international law on intellectual property, which only provides protection to man-made innovations (Brush 2002, 1996; Bragdon and Downes 1998).

Related to these conceptual ideas, a special role is assigned to landraces, which embody a certain degree of human improvement. The local people responsible for the improvement (selection) are vested with some form of claim to payments for their previous efforts. In the policy arena, this claim is formulated in the *farmers' rights* principle (Brush 1992; Frisvold and Cordon 1998).

In contrast to common forms of IPRs, farmers' rights (1) are neither effective in the context of a bilateral exchange of genetic materials nor (2) do they provide for an exclusive individual right to the informational content of genetic material, i.e., a right to control its use by other people. Any compensation payment made according to this principle has a retrospective nature, i.e., it serves to acknowledge the efforts of indigenous farmers in the creation and maintenance of crop genetic diversity (Frisvold and Cordon 1998). Likewise, any right specified in this regard is assigned to a group of farmers defined on a regional, national, or international level. Accordingly, individual claimants need not be identified. If payments are arranged among countries, transaction costs can be reduced relative to an allocation based upon bilateral (private) contracts (Brush 1992; Cooper 2001; Srinivasan 2001).

The farmers' rights principle, similar to the access and benefit sharing (ABS) concept in the CBD, presumes a connection between the incentive for conservation on a local level and the fair distribution of the economic rent associated with plant genetic resources. More precisely, when financial resources are raised among commercial users, farmers are expected to participate in the value of their landraces for commercial breeding. Ideally, the resources received are, in turn, invested in the in situ conservation of traditional farming systems. In contrast to bilateral ABS arrangements, the realization of farmers' rights is currently being discussed as a state-centered approach (Brush 1992, 2002; Cooper 2001).

The farmers' rights principle first emerged on the international agenda in 1989 in the context of a revision of the *International Undertaking on Plant Genetic Resources* (IU) in order to rebalance the agreed upon expansion of exclusive rights on improved genetic material to commercial breeders (Bragdon and Downes 1998).[64] The principle has only recently been reaffirmed in the *International Treaty on Plant Genetic Resources for Food and Agriculture* (IT-PGRFA) (Art. 9 IT-PGRFA).[65] The task of realizing the farmers' rights has been passed on to the national governments (IT-PGRFA Art. 9.2).

On an international level, the IT-PGRFA calls for the implementation of a *financial mechanism* that supports the objectives of the treaty (IT-PGRFA Art. 19.3f).[66] Although the funds provided by this mechanism are effectively

64 The IU represents a nonbinding, international agreement on the conservation and use of plant genetic resources (Bragdon and Downes 1998).

65 The International Treaty on PGRFA was adopted in 1996 and finally entered into force in 2004.

66 As for the CBD, the objectives of the IT-PGRFA confer (1) the "conservation and [(2)] sustainable use" of PGRFA and (3) the "fair and equitable sharing of the

conditional on specific conservation activities, they may help to improve the living conditions of resource-poor farmers in developing countries and may, therefore, be considered indirectly related to realization of the farmers' rights. The financing of this mechanism is, inter alia, based upon mandatory contributions by commercial breeders who use materials from public ex situ facilities and obtain intellectual property protection for the plant variety developed (IT-PGRFA Art. 15.1b(iii)) (Bragdon 2003; Helfer 2005; Liebig et al. 2002).

Property Rights for PGRFA in Ex Situ Facilities

In practice, wild crop genetic resources used in breeding activities are, in most cases, not directly collected in in situ environments. Breeders acquire unimproved material from ex situ facilities (gene banks) in the first place. These facilities provide replications of material from in situ sources, as well as of improved materials that have been used and modified in previous breeding activities (see Section 3.3.2.3).

Regarding the general options to define access and use rights for genetic resources, the stakeholders may agree upon a regime that provides certain *unrestricted access* to the materials in gene banks. The mutual dependency of breeders in the different countries upon a preferably large-scale gene pool may serve as the major argument for such a regime (Cooper et al. 1994). Following the property right concepts introduced above, access without tight restrictions can only be granted to a precisely defined group and is ideally combined with certain extraction and use limitations, as well as conservation rules (Heltberg 2002; Stevenson 1991: 57ff.).[67] These characteristics finally describe the difference between a common property regime and an open access regime.

Concerning a *common property* regime for ex situ genetic resources, an arrangement between the users who have access is needed in order to cover the costs of storing and maintaining the genetic materials.[68] It is commonly agreed that ex situ preservation should include those plant genetic resources currently not in use because of their low use value but that exhibit a positive option value. Since preservation, in this regard, involves positive intergenerational externalities, the arrangement may be implemented on a national level rather than on

benefits arising from their use." This should be obtained "in harmony with the [CBD]" and "for sustainable agriculture and food security" (Art. 1.1 IT-PGRFA) (Linarelli 2004).

67 Considering open access resources and common property goods, it is apparent that access to ex situ facilities can be monitored and controlled much more easily than access to large in situ environments.

68 An agreement between users may arrange an annual fee or levies based upon the material quantities acquired.

the level of firms. Given the relatively unrestricted access, ex situ genetic resources represent public goods that sovereign countries provide internationally.

In contrast to a common property approach, which involves some multilateral coordination, the access to and use of ex situ genetic resources can be regulated in *bilateral contracts* between the private and public institutions of ex situ conservation and the users in the breeding industry. Given the need for access to a large pool of genetic information, a market-based approach to exchange genetic material can incur substantial transaction costs for individual users that affect the profitability of commercial breeding activities and reduce the incentive to develop new crop varieties (Cooper et al. 1994; Ritter and Kosak 1996). Transaction costs can be lowered if bilateral contracts are concluded between governments that act on behalf of their domestic ex situ providers or breeding industry.

Finally, an *IPR* for unimproved crop genetic resources granted to the operators of ex situ facilities is not discussed in the academic and political fora. However, there is a call for international gene banks to represent the interests of developing countries when handing over genetic resources to commercial users. This is because, in the past, developing countries have been the major in situ providers of landraces to the ex situ facilities (Fowler et al. 2001).

With regard to the current policy on the ex situ collections of PGRFA, there is evidence of the pursuit of both the multilateral and the market-based approach: provisions in the framework of the *FAO*, which serves as the major political forum, have historically aimed at a multilateral common property approach (common heritage of mankind). In contrast, the provisions of the *CBD* follow a market-based approach with sovereign rights for genetic resources. Because of these conflicting concepts, the CBD has left aside the regulation of access to and use of PGRFA (Fowler 2000; Bragdon and Downes 1998). The provisions of the recent *IT-PGRFA* attempt to reconcile the two approaches.

The IT-PGRFA, as a binding international agreement, on the one hand, defines and recognizes the sovereign rights of a country over its own PGRFA, as well as its "authority to determine access to the resources that rests with [its] national governments and is subject to the national legislation" (IT-PGRFA Art. 10.1); on the other hand, these sovereign rights are somehow restricted in that countries are committed to the implementation of a *Multilateral System for Access and Benefit Sharing* (IT-PGRFA Art. 10–13). This Multilateral System primarily manifests itself in the global network of ex situ facilities. Its objective is to facilitate access to ex situ genetic resources in a somehow similar way as done in the previous regime of unrestricted access. The difference is that, in addition to this, the mandatory financial mechanism with reference to the farmers' rights and benefit-sharing principles are to be implemented. Because the IT-PGRFA has only recently come into force, both the multilateral system and the financial mechanism are not yet operating formally.

According to the provisions of the IT-PGRFA with regard to the accessing of genetic resources in the Multilateral System, each participating country is obliged to facilitate access to its own ex situ collections in order to have access to the collections of the countries representing the other members of the system. Regarding the financial mechanism, payments should be arranged on a general basis, i.e., without referring to the individual transactions with specific genetic material (Fowler et al. 2001; Helfer 2005; Stoll et al. 2004).[69] This implies that benefit sharing relating to the utilization of PGRFA in the Multilateral System does not refer to the principles of mutually agreed terms (MAT) and prior informed consent (PIC) that apply in the bilateral exchange of wild genetic resources (see Section 3.1.2.1).

Considering the Multilateral System and ex situ facilities included in the System, two issues have attracted public attention:

- first, the *ex situ collections* that are hosted at the international agricultural research centers (*IARCs*) and, thus, are not the subject of national legislation,
- second, the *scope* of the Multilateral System with respect to *crop species* used for breeding and cultivation.

As described in Section 3.3.2.3, the 15 major IARCs cooperate in a formal network, referred to as the *Consultative Group on International Agricultural Research* (CGIAR). Since the CGIAR collections were virtually assembled before the CBD and the IT-PGRFA came into force, their legal status was specified in a contractual agreement between FAO and CGIAR in 1994 (Frivold and Cordon 1998).

According to this agreement, the CGIAR "hold[s] the designated [genetic materials] in trust for the benefit of the international community." The obligation for the CGIAR, in this respect, is to maintain and preserve the ex situ collections, as well as make the genetic material available for all use. By this, it should be secured that genetic resources of the major crops relevant to the world's food production remain in the public domain. In the end, it is intended to make the CGIAR collections part of the Multilateral System (Fowler et al. 2001; Fowler 2000).

With regard to the second aspect, there are *exceptions* to the Multilateral System, i.e., not all PGRFA are subject to this "limited" common property regime: the IT-PGRFA pursues a positive list approach by naming those plant

69 The intention is that commercial breeders who make use of materials provided by the Multilateral System and who have obtained exclusive patent protection for their plant variety should make payments to the financial mechanism (Bradgon 2003; Helfer 2005).

species that are to be included in the Multilateral System.[70] According to Annex 1 IT-PGRFA, 35 food crops and 29 feed crops shall be included in the Multilateral System (Bragdon 2003; Helfer 2005).

Several crop genetic resources, such as soybeans, groundnuts, or sugar cane, are not included in the system. Access to these crops has to be negotiated on a bilateral level with the countries that host these specific plant genetic resources. Consequently, although the IT-PGRFA replaces the legal discrimination of crop genetic resources collected prior to and after the adoption of the CBD, the agreement again provides for a bipartite property right regime, namely for crop genetic resources inside and outside the Multilateral System (Bragdon 2003).

It is argued that this distinction in PGRFA that are easy to access and resources under exclusive access is not a new development: in the previous regime, countries have sometimes enforced exceptions to the principle of unrestricted access and free exchange. This applied in particular to unimproved genetic material of valuable export crops, such as coffee, cotton, oil palm, black pepper, pyrethrum, rubber, or tea (FAO 1998: 284)[71]. Furthermore, a policy of restricted access has often been pursued for highly localized crop genetic resources (Bragdon and Downes 1998).

Intellectual Property Rights and PGRFA

To induce incentives to undertake R&D, exclusive rights on the subsequent use of an information good are assigned to the inventor. Regarding crop genetic resources, intellectual property protection concerns the use of plant varieties. In this respect, the policy on property rights distinguishes use by *breeders* for modification and improvement from use by *farmers* for reproduction. The latter refers to the case where farmers cultivate a protected variety. They withhold a proportion of the resulting harvest (1) for use as seeds for future cultivation or (2) for a reciprocal exchange with other farmers or for an individual market supply ("brown-bag sales") (Swanson 2002; ten Kate and Laird 1999: 125). To preserve the profitability of breeding activities, use by farmers, as well as competing breeders, is restricted by legal provisions.

As an alternative to the legal approach, there have recently been efforts in R&D to transform the biological properties of plant genetic resources in order to inhibit reproduction in the farmer's private domain. These types of crop varieties

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70 It is claimed that, as criteria for the classification of an individual crop in this respect, its importance for "food security," as well as the "interdependence," is considered. Finally, a precise definition of these criteria is not given. Moreover, the current scope of the system is determined in a political process (Fowler 2000).

71 Furthermore, in the past, technical capacity constraints in the maintenance and ex situ reproduction have led to resource scarcity and resulted in (partial and/or temporal) access restrictions (FAO 1998: 284).

are referred to as *genetic use restriction technologies* (GURTs) or, synonymously, "terminator" technologies (Swanson 2002; Srinivasan and Thirtle 2000). Similar to variety-based GURTs (V-GURT), trait-based technologies have also been developed. These T-GURTs allow the inventor to control the subsequent use of the genetic material he has developed by means of biotechnological keys of which he is in exclusive possession (Swanson 2002; Stoll et al. 2004).[72] These GURTs imply that breeders no longer rely upon the enforcement of legal IPRs in order to appropriate the value added to their inventions. However, these technologies are currently subject to controversial debates, since it is claimed that they have certain undesirable effects upon competition on markets for varieties and the international distribution of wealth among (resource-poor) farmers and multinational breeders (for more details, see Srinivasan and Thirtle (2003), Stoll et al. (2004), Swanson and Goeschl (2002), or UNEP/CBD (2002b)). Since GURTs are not yet broadly used, there is a need for the intellectual property protection of crop varieties.

Generally, an IPR system for PGRFA has to address two major issues:

- first, the choice of an appropriate *form of IPR* that addresses the particularities of PGRFA,
- second, the interplay between the use of *unimproved genetic information in the public domain* and exclusive IPR on improved or invented information.

Regarding the design of an IPR for PGRFA, it is claimed that an effective IPR system should, on the one hand, create sufficient incentives for breeders to undertake breeding activities in such a way that new, promising plant varieties are realized; on the other hand, any exclusive rights should take into account the strong externalities in breeding activities, i.e., breeders should be allowed to use protected materials for modification and further improvement.[73]

The latter argument suggests that extensive rights for an individual breeder to control the subsequent use of his invented seed product may not represent the appropriate form of IPR, since it can impede the gene flow into the breeding processes and, thereby, lead to a decreasing R&D output. In contrast, strong control rights for the inventor do not imply that genetic materials are retained completely: protected material can be exchanged on the basis of a licensing agreement, i.e., a bilateral contract on the right to use improved genetic material for precisely defined research purposes (ten Kate and Laird 1999: 145ff.;

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72 This is obtained by making the effectiveness of an innovative trait dependent upon the application of an "initiator," i.e., a specific complementary material that, in turn, is in the exclusive possession of the inventor (Swanson 2002; Stoll et al. 2004).

73 Furthermore, there are political demands to allow farmers to generate seeds for their own use because they made an essential contribution to the world gene pool through their own crop selection in the past.

Nottenburg et al. 2003). To conclude, there are arguments for and against strong IPRs, namely patents for PGRFA. In general, it is possible to apply different forms of IPRs. In addition to patent rights, copyrights, or trade secrets, a sui generis IPR system that addresses the particularities of PGRFA can be defined (FAO 1998: 395ff.).

In practice, the IPR system for improved crop genetic resources has been subject to changes in recent decades that have been driven by two different international regimes: the TRIPS agreement and the *International Union for the Protection of New Varieties of Plants* (UPOV).

When modern breeding technologies were first developed in the late nineteenth and early twentieth century, crop varieties were part of the worldwide system of unrestricted access and breeding activities were predominately undertaken by the public sector. In the second half of the twentieth century, initial steps on use restrictions and internationally recognized exclusive rights were agreed upon in the forum of the UPOV. This intergovernmental organization was established as a consequence of the 1961 International Convention for the Protection of New Varieties of Plants (Bragdon and Downes 1998; Swanson and Göschl 2000).

A core instrument of this arrangement is the plant breeder's right (PBR) as a sui generis IPR form for plant varieties.[74] A PBR granted for a plant variety is mutually recognized within the UPOV member countries. Regarding the use of a plant variety by farmers and competing breeders in the system of PBRs, the UPOV countries have implemented fewer use restrictions for farmers and competing breeders. This has been manifested in the principles of (1) the *farmer's privilege*, which allows farmers to withhold seeds for replanting and (2) the *breeder's exemption*, which provides breeding organizations with a general freedom to operate (FAO 1998: 397; Dutfield 1999; IPGRI 1999).

Over the last decades, however, several revisions of the UPOV have arranged for a step-wise reallocation of property rights in such a way that the exclusive rights for the breeder who has developed a variety are reinforced, while the farmer's privilege and the freedom to operate are partly pushed back (Frisvold and Cordon 1998; Dutfield 1999; Acharya 1991). According to provisions in international law, the UPOV member countries have some freedom of choice in defining the scope of rights for plant varieties. This includes the fact that the right for farmers may be preserved, as well as that patent protection for varieties is enabled (Bragdon and Downes 1998).

The members of the UPOV initially consisted of developed countries. Nevertheless, Figure 8 illustrates that, in the last decade, more and more developing countries have acceded to the UPOV. Of the current 58 members, nearly two

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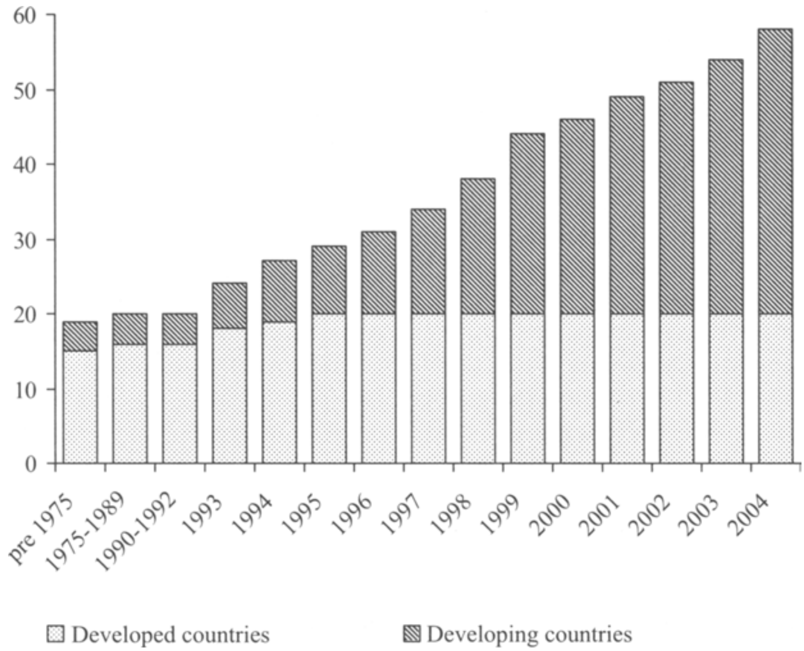
74 The term "plant variety protection" (PVP) is used synonymously to mean PBR.

Figure 8:
Countries as Members of the UPOV



Source: UPOV (2005), own representation.

thirds represent developing countries and countries in transition. For several of them, there is evidence that industrial breeding sectors with increasing investment in R&D have been established. This implies that these countries increasingly derive benefits from securing IPRs for registered plant varieties (Goeschl and Swanson 2000).

This increase in the number of members of the UPOV can be seen in the political context of the obligations arising from the TRIPS agreement: the UPOV and its underlying international convention finally represent a sector-specific international IPR regime that is compatible with the TRIPS agreement, which applies to information goods in general. Although the TRIPS agreement calls for countries to provide patent protection for biotechnological goods and processes (Art. 27(1) TRIPS), a specific regulation is arranged for plants (see Section 3.3.2.2): signatory countries to the TRIPS agreement can choose between either allowing patent rights for plant varieties or implementing a sui generis system.[75] Without referring explicitly to the PBRs, the existing UPOV regime is generally considered a property right system that fits into this context (IPGRI 1999; Bragdon and Downes 1998; Brush 1992). In total, the development of the TRIPS agreement and the revision of the UPOV regime imply that commercial seed and plant varieties are increasingly subject to IPR protection (Frisvold and Cordon 1998).[76]

With regard to efficiency considerations, the literature on IPRs for PGRFA argues that the international property rights regime only creates incentives to invest "at the end of the industry," i.e., in breeding activities, but not "in the earlier parts of the industry." These early stages relate to the preservation of the world's gene pool, including unimproved genetic material in in situ conditions (Swanson and Göschl 2000).

Furthermore, against the background of *equity considerations*, the international regime is often regarded as biased against the unimproved materials that frequently originate from developing countries, while intellectual property protection of plant varieties, i.e., material in an improved and modified form, is (still) predominately granted to breeders in the developed countries (Frisvold and Cordon 1998; Lerch 2000).

In this context, there are controversies as to whether and when the improvement of plant varieties should actually be rewarded with the granting of an IPR (Dutfield 1999). Sometimes IPRs are granted to crop varieties that are closely related to landraces. If these IPRs were effectively enforced, the *traditional use* of unimproved material could be restricted. Resource-poor farmers in developing countries who strongly depend on the crops available would be affected by this in particular, which would ultimately come into conflict with common concerns of equity (Bhat 1996; Pardey et al. 2003).

Aside from the restriction of traditional use, it is often claimed that IPRs may impede the use of materials that are contained in the ex situ collections but belong to the global *public domain*. This concerns particularly those genetic resources that are stored in the gene banks of the CGIAR (Fowler et al. 2001; Acharya 1991).

As a strict means of protecting the gene pool in the public domain, there is a public discussion whether to ban IPRs for varieties that rely upon genetic materials from these international gene banks, particularly samples of landraces that are in use in traditional farming systems. However, this would more than

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75 Currently, the use of patents to protect plant varieties is implemented in Australia, Japan, and the United States (Fowler et al. 2001).

76 The recent IT-PGRFA addresses IPRs to plant varieties with regard to materials provided within the Multilateral System (Helfer 2005; Stoll et al. 2004).

likely divert the breeders' demand for ex situ genetic resources to other sources, such as in-house collections or private gene banks. This is because they cannot expect to capture the value of their breeding efforts on the market for commercial seeds without an IPR. Furthermore, if in this respect the breeders cannot draw on other sources to obtain genetic material of comparable quality, an IPR ban may cause fewer crop varieties to be developed.[77] If for this reason, an IPR for plant varieties is not excluded, the way exclusive rights should be specified needs to be defined (Helfer 2005).

Considering *PBRs*, the granting of this form of IPR for a plant variety requires a set of conditions to be fulfilled. Namely, a new variety has to be (1) distinctive, (2) uniform, and (3) stable. While these conditions virtually exclude any IPR to landraces for traditional users and, thus, do not address incentives for in situ preservation, they also impose limits on the unjustifiable approval of the intellectual property protection of a crop variety. Furthermore, even if a breeder is awarded a PBR, many uses by other breeders for research purposes are not excluded (Dutfield 1999; Fowler et al. 2001). Consequently, the granting of PBRs for varieties whose development is based on the ex situ resources in the public domain does not impede the further distribution of the resources.

Considering potential restrictions resulting from *patents* that enable a stricter control of use by other people than that of PBRs, it is shown that, in current national patent laws, exclusiveness typically refers to the use of patented genes that have been isolated and purified by the patent holder.[78] Current systems of patent rights, however, do not arrange for exclusive rights to all use of materials naturally containing the patented gene. Accordingly, it is argued that genetic material belonging to the public domain is currently in less danger of being privatized. Nevertheless, it will be, in principle, possible to implement or revise national laws on patents in the future in order to arrange for more extensive rights for the patent holder (Fowler et al. 2001; IPGRI 1999).[79]

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[77] Furthermore, since it is intended to make private contributions to the financial mechanism of the IT-PGRFA conditional on the market revenues the commercial breeders receive, a lack of intellectual property protection constrains the attainable revenues and, therefore, also the funds available for the preservation of PGRFA (Helfer 2005).

[78] Currently, only certain countries, such as Australia, Japan, and the United States, implement patent protection for plant variety (Fowler et al. 2001).

[79] Furthermore, it is argued that patent protection is awarded on a national level and, therefore, does not a priori confer worldwide property rights, even though the TRIP agreement calls upon its signatory countries to implement a national IPR system with regard to plant varieties. In effect, each national government has to approve a patent right within its country. Otherwise, the genetic material is freely available for research purposes (Fowler et al. 2001; Pardey et al. 2003).

Aside from legal limits to an IPR for ex situ genetic material that has previously been part of a system of limited common property regime, its removal from the public domain may be ruled ex ante on the basis of contractual arrangements. This refers to *material transfer agreements* (MTAs), which regulate the exchange of genetic resources between provider and user (FAO 1998: 402ff.; Barton and Siebeck 1994).

MTAs have primarily been used to arrange the transfer of improved genetic material. This type of contract is increasingly used by facilities that store ex situ materials and transfer material for commercial users. MTAs specify the right of the material recipient to apply for intellectual property protection for the new product he has developed through the use of the material received. For the ex situ gene banks of the CGIAR, the policy on IPR was first addressed in the 1994 FAO-CIGAR agreement. The principles stated therein have been affirmed in the IT-PGRFA, which aims to rule out cases where the use of materials in the public domain is restricted by IPRs afterwards (Helfer 2005; OECD 1996; Stoll et al. 2004).[80]

Regarding the content of an MTA, the implementation of Art. 12.3(d) IT-PGRFA, which states that recipients "shall not claim any intellectual property or other rights that limit the facilitated access to the [PGRFA], or their genetic parts or components, in the form received from the Multilateral System," is crucial. Given the general reflections on the requirements for intellectual property protection, it is supposed that the implementation of this provision will become a major issue in the future (Helfer 2005; Linarelli 2004).

Animal Genetic Resources in the Agricultural Sector

Beside PGRFA, the management of genetic material for animal breeding processes also attracts attention (Gollin and Evenson 2003; Mendelsohn 2003; Roosen et al. 2003). The issue of AnGR displays some similarities with the issue of PGRFA. For example, threats to animal genetic diversity are partly due to the same factors as for PGRFA, namely the displacement of indigenous breeds by modern and more productive ones. Also, the FAO is considered to be the relevant international forum to address this issue. In contrast to plant genetic resources, the management of genetic material seems to be largely hosted in ex situ conditions, i.e., breeds from domesticated animals are used. The linkage to natural or less modified ecosystems remains unclear. Furthermore, the role of

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80 The privatization of public resources in agricultural research is a complex topic. In practice, private breeding organizations often depend upon the technologies and resources that the public breeding organizations possess and vice versa. There is evidence of increasing private-public cooperation in this field. Consequently, privatization may occur when actors in the public sector exclusively provide genetic material or associated knowledge—with or without addressing IPRs (Stoll et al. 2004).

cross-border trade in breeds for biotechnological applications has not been investigated. Moreover, there is evidence that, in order to implement an appropriate management regime, the focus falls upon the local or regional scale.

After all, international policies in the area of AnGR are just developing. For example, current activities aim at the description of the state of the world's AnGR as a prerequisite for further research and policy design. For these reasons, the issue of AnGR is not investigated any further in the remainder of this study.

To resume, the analysis of property rights for genetic resources presented in this section has focused upon a discussion of the functional role of these property right institutions and a brief description of their practical implementation. Little is said about why the existing property rights regimes were created this way. An analysis of this question needs to address aspects of institutional change and collective actions (among countries on an international level and governmental and private stakeholders on a national level) (Heltberg 2002; Olson 1965). Furthermore, aspects of political economy need to be addressed in more detail and, in this respect, particularly the conflicting views of developed and developing countries with respect to the access and use of unimproved and improved genetic materials (Görg and Brand 2000). In the literature, it is argued that the actual property right institutions may be subject to inefficiency as a result of "inertia, friction, vested interests and collective action problems" that, in the end, distort the process of institutional change and lead to the suboptimal allocation of resources (Heltberg 2002). Although I briefly address some of these aspects in Section 3.2.2, a detailed investigation of the institutional change of property rights for genetic resources lies outside the scope of this study.

3.2 Commercial Uses of Genetic Resources and Biodiversity Conservation: Theory and Practical Problems

Property rights for genetic resources are a prerequisite for market creation and, therefore, for the creation of market-based incentives for biodiversity conservation. The question is how trade in genetic resources contributes to the conservation of biodiverse ecosystems and, thereby, the provision of valuable ecosystem services (see Section 2.3.3).

As mentioned earlier, the special features of ecosystems are the following: first, *multiple outputs* in the form of ecosystems services of varying private and public good properties and, second, complementarity or *connectedness* as output bundles. The composition of these bundles depends upon site-specific ecological conditions and human land use. An undesirable decline in the flow of ecosystem services occurs due to (1) the modification of natural areas into managed ecosystems with less biodiversity and (2) nonsustainable resource depletion (see sections 2.1.1, 2.3.1, 2.3.3). Focusing upon ecosystem services where excludability can be established, the question is whether and to what extent specifying and enforcing property rights for these services and their exchange on the market creates incentives for private efforts in conservation. Given the multiple valuable ecosystem services and their connectedness, trade in the ecosystem services that represent private goods assists the maintenance of public-good-like services for which no market comes into existence.

In this regard, attention is drawn to genetic resources: if commercial users engaged in R&D show an interest in genetic information from in situ environments, a private provider may generate income from preserving natural habitats as bioprospecting areas. The payments received compensate for forgone revenues from alternative land use and, therefore, render preservation the more profitable form of land/resource use.

In the following, I use two simple partial equilibrium models to describe how the linkage between the markets for genetic resources and biodiversity preservation is established in an idealized market environment. In Section 3.2.2, I briefly enumerate impediments to an effective linkage by drawing upon the analysis summarized in Figure 5.

3.2.1 Joint Supply of Private and Public Ecosystem Services: The Theoretical Concept

As demonstrated in Section 2.3.1, the management of biodiversity and ecosystem services can be conceptualized as an economic problem of choosing (1) the optimal form of land use or (2) the optimal path for resource extraction. In this regard, I introduce two simple models that describe ecosystem changes as *land use change* and *natural resource extraction*. Although genetic resources are generally allocated in different institutional frameworks, both modeling frameworks assume that genetic resources are traded as private goods between in situ providers and commercial users.

A Simple Static Model of Alternative Land Use and Joint Supply

First, I develop a simple static model of alternative land use where genetic resources can be collected on preserved natural lands and sold on the market. For the conceptualization of land use and markets for land-based commodities, studies on the theory of public goods, externalities, and joint supply (e.g., Buchanan 1966; Heal 2003; Holm-Müller 1999) provide helpful insights. I also consider more recent applications of the theory to the topic of multifunctional agriculture (Romstad et al. 2000; Blandford and Boisvert 2002; Peterson et al. 2002; Paarlberg et al. 2002; Lankoski and Ollikainen 2003).

Given the diverse economic nature and multiple characteristics that genetic material and genetic information can display in practice, the model assumes that land property rights and property rights to biological materials hosted on the land are perfectly enforced and assigned to the private landowners. Commercial users in R&D only gain possession of genetic information if they acquire genetic material from in situ conditions that are represented by private land in a preserved and relatively undisturbed state.

For describing the *property rights* for genetic resources and the revenues the resources generate in the market, let us distinguish three stylized scenarios:

- Property rights for genetic resources are not specified, i.e., the resources are subject to a (de facto) open access regime.
- Property rights for the resources are specified and enforced. The owners are only paid for the resource value. No arrangements for the sharing of benefits from genetic information are made.
- In addition to well-specified and well-enforced property rights, benefit sharing arrangements are obligatory. Accordingly, the resource owners receive payments above the direct resource value.

In the first scenario, there is no incentive for conservation and sustainable resource use. In the second and third scenarios, the private incentive to conserve depends upon the competition between in situ providers and the negotiation of a benefit sharing arrangement, which effectively determine the providers' revenues (see Section 3.1.2.1).

The model assumes a *positive market price* for genetic resources: depending upon the relative prices, a provider can earn sufficient revenues from a market supply of genetic resources and, thereby, has an incentive to preserve ecosystems.[81] Consider an individual representative ecosystem in a static one-period setting. A natural area whose size is normalized to 1 represents the ecosystem. At first, the area shows a high level of biodiversity.

To model alternative land uses and biodiversity loss, let us define that the area can be converted and used in the production of an agricultural good as, q_y, whereas the degree of biodiversity is assumed to decline drastically with the consequence that no other valuable ecosystem services are provided.

Alternatively, the natural area is preserved and/or placed under a sustainable management regime. In this case, (1) a bundle of valuable but intangible public-good-like ecosystem services, q_e, is generated and (2) genetic resources, q_g,

[81] For simplicity, I assume that no intermediaries take part in the trading so that the commercial users pay the landowner directly.

are preserved in situ. Let us assume that genetic resources can be collected without affecting the flow of the intangible ecosystem services. Therefore, the two goods represent joint products of the preservation.

Let α, $0 \le \alpha \le 1$ denote the proportion of land that is *preserved* and $1-\alpha$ the proportion in *agricultural use*. An individual profit-maximizing landowner decides upon the allocation of land. Furthermore, suppose that the extraction/ appropriation of the three goods demands some *additional management input* (labor and technology). The landowner has to decide upon the allocation of land and how much input to purchase for each of the production activities.

To describe the *decision-making process* analytically, we choose a sequential recursive approach. In the initial stage, the optimal allocation of land, α^*, is determined given the optimal quantities of input in the different productions x_y^*, x_e^*, x_g^*, which are determined in the second stage. To describe the optimal choice of input, the proportion α is initially considered as given, as are the vectors of good prices, p_e, p_g, p_y, and input prices, $p_{x_e}, p_{x_g}, p_{x_y}$. The profit-maximization problem for the *preserved area* is given as

$$(3.1) \qquad \max_{x_e, x_g} \pi_p(\overline{\alpha}) = \max_{x_e, x_g} p_e q_e(\overline{\alpha}, x_e) + p_g q_g(\overline{\alpha}, x_g) - p_{x_e} x_e - p_{x_g} x_g \,.$$

That of the *converted area* is represented by

$$(3.2) \qquad \max_{x_y} \pi_c(\overline{\alpha}) = \max_{x_y} p_y q_y(1-\overline{\alpha}, x_y) - p_{x_y} x_y - c(1-\overline{\alpha}) \,.$$

In addition to the direct cost of agricultural cultivation, the landowner has to carry the conversion costs described by the function $c(1-\alpha)$. Using the first-order conditions for a profit maximum, the optimal input quantities can be derived, where each solution is a function of α, the final good prices, and the input prices.

Based upon these results, the profit-maximization problem in the first stage can be solved for the optimal land allocation:

$$(3.3) \qquad \max_{\alpha} \, \Pi(\alpha) = \max_{\alpha} \pi_p(\alpha, x_e^*(\alpha), x_g^*(\alpha)) + \pi_c(\alpha, x_y^*(\alpha)) \,.$$

Using this approach and solution for α, optimal allocations of land, α^*, are represented as a function of the prices for the final goods and the input prices, including the costs of conversion. The uniqueness of the optimum depends upon the quasi-concavity of the underlying production functions.

To assess the allocation with regard to *efficiency*, the demand side of the goods considered, i.e., the utility derived therefrom, needs to be defined. The conditions for Pareto optimality in such a general equilibrium framework typically refer to the identity of the marginal rates of substitution between the individual goods and a numéraire on the one hand and the corresponding marginal rate of transformation on the other. Externalities, including joint supply, lead to modifications in these identity conditions (Paarlberg et al. 2002; Buchanan 1966).

Since market prices do not completely capture the social value of genetic resources (Lerch 1994; von Amsberg 1995), the level of conservation in a pure market allocation is below the socially optimal level. To study efficiency, household utilities and the demand side of genetic resources have to be specified in detail, which would add further complexity to the model. Instead, let us continue with the assumption of exogenous resource prices. The aim is to illustrate how changes in relative prices influence the level of conservation in the market equilibrium. For this purpose, let us use a simple example with functional forms for the production and cost functions.

The Land Use Model: An Example with Functional Forms

For simplicity, let us define that $p_g > 0$ but that $p_e = 0$: the owner of preserved natural land only receives market revenues from the supply of genetic resources. To specify the production of genetic resources and the agricultural good, let us use Cobb–Douglas functions:

$$(3.4) \quad q_g = \alpha^{1-\gamma} x_g{}^{\gamma},$$

$$(3.5) \quad q_y = (1-\alpha)^{1-\rho} x_y{}^{\rho}.$$

For given levels of land allocation, α^*, let us use the profit functions in equations (3.1) and (3.2) in order to derive the first-order conditions for a profit maximum in both the preserved and converted part of the land. These conditions are described by the identity of the marginal product of the management input and the input price:

$$(3.6) \quad p_g \frac{dq_y}{dx_y} = p_g \gamma \alpha^{1-\gamma} x_g{}^{\gamma-1} = p_{x_y},$$

$$(3.7) \quad p_y \frac{dq_y}{dx_y} = p_y \rho (1-\alpha)^{1-\rho} x_y{}^{\rho-1} = p_{x_y}.$$

Given the definition of production functions, the second-order conditions are satisfied. By transforming the first-order conditions, the optimal inputs can be derived in both productions:

(3.8) $$x_g^* = \alpha p_{x_g}^{\frac{1}{\gamma-1}} \left(\gamma p_g\right)^{\frac{1}{1-\gamma}},$$

(3.9) $$x_y^* = (1-\alpha) p_{x_y}^{\frac{1}{\rho-1}} \left(\rho p_y\right)^{\frac{1}{1-\rho}}.$$

Inserting the optimal input into the corresponding profit functions, yields the profits in both land use regimes for a given level of α. The profits in preservation are

(3.10) $$\pi_p(\alpha) = \alpha p_g^{\frac{1}{1-\gamma}} p_{x_g}^{\frac{\gamma}{\gamma-1}} \gamma^{\frac{1}{1-\gamma}} \left(\frac{1}{\gamma} - 1\right).$$

Turning to agricultural use, let us assume that the costs of conversion are positive and increasing:

(3.11) $c(1-\alpha) = (1-\alpha)^{\varepsilon}$ with $\varepsilon > 1$.

The profits in the agricultural production are

(3.12) $$\pi_c(\alpha) = (1-\alpha) p_y^{\frac{1}{1-\rho}} p_{x_y}^{\frac{\rho}{\rho-1}} \rho^{\frac{1}{1-\rho}} \left(\frac{1}{\rho} - 1\right) - (1-\alpha)^{\varepsilon}.$$

As shown by equation (3.10) the profit in the preserved area changes linearly in α: when α is increasing, the profit in the preserved area increases proportionally. Let the profit of a marginal unit of preserved land be denoted by

(3.13) $$\beta_p = p_g^{\frac{1}{1-\gamma}} p_{x_g}^{\frac{\gamma}{\gamma-1}} \gamma^{\frac{1}{1-\gamma}} \left(\frac{1}{\gamma} - 1\right).$$

When neglecting the cost of conversion for a moment, equation (3.12) implies that the profit in the converted area also changes linearly in α, but decreases proportionally. Let the (gross) profit of a marginal unit of converted land be denoted by

(3.14) $$\beta_c = p_y^{\frac{1}{1-\rho}} p_{x_y}^{\frac{\rho}{\rho-1}} \rho^{\frac{1}{1-\rho}} \left(\frac{1}{\rho} - 1\right).$$

Referring to equation (3.3), the landowner's profit-maximizing choice of land use reduces to

$$(3.15)\quad \max_{\alpha}\ \Pi(\alpha) = \alpha\beta_p + (1-\alpha)\beta_c - (1-\alpha)^{\varepsilon}.$$

Solving for the profit maximum yields

$$(3.16)\quad \alpha^* = 1 - \left(\beta_c - \beta_p\right)^{\frac{1}{\varepsilon-1}} \varepsilon^{\frac{1}{1-\varepsilon}}.$$

It can be shown that α^* represents an interior solution whenever $\varepsilon > \beta_c - \beta_p \Leftrightarrow \alpha^* > 0$ and $\beta_c > \beta_p \Leftrightarrow \alpha^* < 1$. Given this result, it is of interest to note how an increasing price, p_g, influences the allocation. Simple comparative statics shows that it increase the size of the preserved area (with the range of an interior solution):

$$(3.17)\quad \frac{d\alpha^*}{dp_g} = \frac{\varepsilon^{\frac{1}{1-\varepsilon}}}{\varepsilon-1}\left(\beta_c - \beta_p\right)^{\frac{2-\varepsilon}{\varepsilon-1}} p_g^{\frac{\gamma}{1-\gamma}} p_{x_g}^{\frac{\gamma}{\gamma-1}} \gamma^{\frac{1}{1-\gamma}} > 0.$$

Note that β_c and β_p are functions of the prices and control the magnitude of the effect of increasing price for genetic resources. This exogenous price of genetic resources in this model framework represents the commercial users' willingness to pay on the demand side. Whenever this willingness to pay happens to be low relative to opportunity costs of preservation on the supply side, the impact of the trade in genetic resources is of minor importance, i.e., the size of the preserved area tends to be zero.

The model illustrates how the market for genetic resources can function as a means to induce incentives to preserve biodiversity and how it is influenced by the interplay between the costs of preservation and the willingness to pay. In practice, these two determinants are influenced by many factors, whereas the importance of the individual factors and the interactions between them can vary considerably between sites and over time. Accordingly, the potential impact of the market incentives for conservation may only be assessed reasonably in a site-specific context. While the opportunity costs of preservation depend inherently upon the local economic and ecological conditions, the industrial commercial users' willingness to pay in particular is driven by factors on international markets. Accordingly, I attempt to identify certain general trends concerning this and, thereby, support certain general implications. The empirical analyses in sections 3.3 and 3.4 develop this aspect.

A Simple Dynamic Model of Resource Management and Joint Supply

Since the nature of biodiversity conservation is inherently dynamic, I provide an analysis of a joint supply of ecosystem services and trade in genetic resources in the framework of intertemporal resource management. I introduce a simple dynamic model that describes how an increasing market price for genetic resources can increase the incentive to preserve natural habitats.

The same assumptions on property rights apply as in the static case. The owner of biodiversity now manages natural habitats as a renewable resource. For the analytical representation of the decision on resource management, the tools of dynamic optimization used in natural resource economics are applied (van Kooten and Bulte 2000: 218ff.; Siebert 1983: 110ff.).

Let the initial resource stock be denoted by r_0. Through resource extraction, y_t, the landowner obtains a private good, which yields a market, p_y. c_y denotes the unit cost of extraction. Alternatively, the preserved resource stock generates private and public goods. Suppose private goods are genetic resources that yield a given market price, p_g, per unit and let the relationship between the provision of genetic resources and preservation be described by the function $h(r_t)$. As will be shown, the results crucially depend upon the functional form. I will discuss the properties of the function below. Suppose that in situ genetic material is only extracted in very small quantities so that the overall resource stock is not reduced. The assumption of small extraction quantities is justified by the empirical evidence on the use of genetic resources as an information good (see Section 3.1.1). Furthermore, the resource owner does not receive any return from public ecosystem services that are jointly supplied by preservation. c_p denotes the unit cost of preservation that the resource owner incurs in order to inhibit illegal extraction or the detrimental impact of exotic species. The resource owner's optimization problem is to maximize profits over time, $t = 1..\infty$, discounted by the rate δ, with $0 < \delta < 1$.

The constraint on optimization is represented by the regeneration capacity of the resource base. It is given by a conventional logistic growth function subtracted by the resource extraction, y_t, in each period, t.

$$(3.18) \quad \max_{y,r} \int_0^{\infty} \left((p_y - c_y) y_t + p_g h(r_t) - c_p r_t \right) e^{-\delta t} dt$$

$$\text{s.t.} \quad \frac{dr}{dt} = \dot{r} = f(r_t) - y_t \, .$$ [82]

Based upon this, the current value Hamiltonian is formulated as

82 Regarding the growth function, it holds that

$$\frac{df(r)}{dr} > 0 \; for \; 0 < r < \bar{r} \, , \; \frac{df(r)}{dr} < 0 \; for \; \bar{r} < r < \hat{r} \, , \; \frac{df(r)}{dr} = 0 \; for \; r < 0; r = \bar{r}; r > \hat{r} \, .$$

$$(3.19)\quad H_c = (p_y - c_y)y_t + p_g h(r_t) - c_p r_t + \varphi(f(r_t) - y_t).$$

In the following, let us omit the time argument. Thus, for example, y_t is equivalent to y. The necessary conditions for the maximum principle are then derived by the optimality condition

$$(3.20)\quad \frac{dH_c}{dy} = 0 \Leftrightarrow p_y - c_y = \varphi,$$

and the costate condition

$$(3.21)\quad \dot{\varphi} - \delta\varphi = -\frac{dH_c}{dr} \Leftrightarrow \dot{\varphi} = -p_g \frac{dh(r)}{dr} + c_p - \varphi \frac{df(r)}{dr} + \delta\varphi.$$

These equations finally determine a system that, if sufficiency conditions are satisfied, describes the optimal solution. Let us neglect the optimal extraction path but concentrate on the long-term equilibrium. The steady state requires that $\dot{\varphi} = 0$. Using the equations for the optimal solution, the steady state requires that

$$(3.22)\quad \delta = \frac{df(r)}{dr} + \frac{p_g \frac{dh(r)}{dr} - c_p}{p_y - c_y}.$$

This equation is similar to the well-known *arbitrage relationship* between the payoffs from extraction represented by the discount rate and the payoffs from preservation represented by the natural resource growth (Siebert 1983: 114). In comparison to the conventional definition of this arbitrage equation, the net returns generated from the provision of genetic resources, weighted by marginal opportunity cost, are added to the payoffs of preservation.

Let us formally analyze how an increasing market price for genetic materials, $p_{g,}$ influences the private decision to preserve the natural resource stock. For this purpose, let us consider the long-term equilibrium, r^*, a benchmark. Applying the implicit function rule to the steady state of the optimal solution yields

$$(3.23)\quad \frac{dr^*}{dp_g} = -\frac{\frac{dh(r)}{dr}}{(p_y - c_y)\frac{d^2 f(r)}{dr^2} + p_g \frac{d^2 h(r)}{dr^2}}.$$

This equation shows that the functional form of $h(r)$ matters for the assessment of the impact of the market for genetic resources. However, a reliable empirical relationship between the resources that are of interest to R&D and the size of the natural resource base, r, as defined here, cannot be identified on a reasonable basis. Only the species-area relationships investigated in ecology can support certain implications in this regard: empirical estimates of these relationships imply decreasing returns to scale of units of the natural habitat in the provision of species richness (Armsworth et al. 2004). Furthermore, findings in ecology show that evolutionary dynamics, i.e., the generation of additional genetic diversity, may be more inert in protected and preserved ecosystems than in ecosystems that display a certain degree of external disturbance. Based upon these considerations, it is reasonable to assume that the first derivative of $h(r)$ is nonnegative. The second derivative is likely to negative or zero, but not positive.

Equation (3.23) implies that if in situ genetic diversity is virtually nonsensitive to changes in the natural resource base, r, the first derivative of $h(r)$ in the nominator is (close to) zero; an increasing relative scarcity of genetic resources represented by an increasing market price, p_g, does not induce a higher level of preservation, r^*. In contrast, if some sensitivity prevails, i.e., the first derivate is positive, an increasing market price for genetic resources induces a higher level of preservation whenever the second derivative of the logistic growth function, $f(r)$, is negative, which is typically satisfied.

Before continuing, some caveats need to mentioned. The model assumes a price for genetic resources, p_g, that remains constant over time. In other words, it implies that there is an ongoing industrial demand for genetic material originating from the resource base considered. As argued in Section 3.1.1, this may not be true for many commercial uses of genetic resource since the information embodied can be replicated in ex situ conditions.

In the model, such a scenario would affect demand and lead to a decline in the resource price over time. In modification of this thought, I may also assume that the resource owner can only realize a positive price in one specific period, T, with $1 < t = T < \infty 1$. Prior to this period, commercial users are unaware of the value of the in situ genetic material. After they collect all of the relevant in situ genetic material and store it in ex situ facilities, the demand for material ceases and the price drops again to zero. I neglect the formal representation of these two cases here, since the direction of the supportive impact of the trade in genetic resources on conservation remains the same.

In addition, more complex representations of the ecosystem dynamics may be incorporated in the model. As a consequence, more volatile optimal extraction paths or even unpredictable long-term outcomes may occur (e.g., Perrings and Walker 2004). I also leave these aspects for future research.

Before considering the empirical information on prices and quantities traded on the market in sections 3.3 and 3.4, I briefly review the practical problems that impede efficiency, as well as the effectiveness of trade in genetic resources.

3.2.2 Practical Problems of Bioprospecting

Contractual arrangements on transactions with wild genetic resources are often summarized under the term "*bioprospecting*" (see Section 3.3.2.2). Given ideal conditions, in particular with respect to the specification and assignment of property rights for wild genetic resources, bioprospecting leads to the efficient allocation of these economic goods (Coase 1960; Siebert 2005: 99ff.). By applying the literature on transaction cost economics, reasons can be identified as to why the outcome of the bilateral bargaining on these genetic resources may not be efficient (Williamson 1998).

Transaction cost economics, in turn, implicitly assumes that the *institutional environment* of a market for genetic resources is already well defined (Williamson 1998). However, this prerequisite does not hold in practice. Moreover, on an international level, the use of genetic resources and their embodied information is regulated by multiple regimes in the fora of the CBD, FAO, and WTO, with the consequence that there is considerable overlap, as well as conflicts between these different regimes (Görg and Brand 2000; see also Section 3.1.2). Regarding the national level, it has been observed that developing countries providing genetic resources often lack consistency in their legal frameworks and procedures, which is accompanied by a lack of coordination among the parties involved (Dhillion et al. 2002). Sometimes, missing legal frameworks that regulate the equitable sharing of benefits derived from genetic resources are considered a major obstacle to creating effective incentives for conservation (Dhillion et al. 2002). In other cases, excessive bureaucratic regulations in these countries seem to divert the demand for R&D input away from in situ genetic material, with the consequence that less revenues are obtained from supplying genetic resources and less money is invested in in situ conservation (ten Kate and Laird 1999: 297ff.).

In more detail, property rights are often specified incompletely but there is also frequently a discrepancy between de jure and de facto rights. This is due to practical constraints in enforcing the specified property rights. Problems of enforceability can occur with regard to rights to land property and biological resources, as well as those to improved genetic material or genetic information.

For example, landowners are unable to control access to in situ habitats in such a manner as to rule out the unauthorized acquisition of genetic resources. The enforcement of property rights to genetic resources is generally complicated by the fact that these resources are easy to transport and replicate (Frisvold and Cordon 1994).

Furthermore, even when individual landowners on a local level agree to hand over genetic material to commercial users, the transaction may violate access and benefit sharing provisions on a national level. Such deals constitute illegal acquisition, since, by transferring the physical material, the user obtains the de facto use rights to both the material and the information embodied in it. It is difficult for the resource country to enforce any further claim to the material or information. Consequently, although national access and benefit regulations are in place, the resource countries often cannot prevent such forms of "biopiracy."

Finally, there may be infringements of IPRs for biotechnological products, such as unauthorized use for replication or use in further research, which affect the innovator's ability to appropriate the value of his innovation (Giannakas 2003). As far as these new product developments rely upon the input of genetic resources in R&D, IPR infringement can lead to the stagnation of R&D efforts and, therefore, the demand for genetic resources. This, in turn, leads to low revenues for suppliers and relatively small returns from biodiversity preservation.

Transaction Costs and Vertical Integration

Imperfections in specifying and enforcing property rights hinge upon the existence of transaction costs that restrict an efficient bargaining solution. In the literature, numerous definitions of transaction costs exist.[83] Generally, the focus in identifying and describing transaction costs can be placed more narrowly upon market transactions and, therefore, the costs incurred by the stakeholders on the market. Considering transactions with genetic resources, these costs are particularly the costs of exclusion and the costs of communication and information (Brush 1996, 2002). They increase the supply costs of genetic resources or reduce the net benefit on the demand side. In this respect, transactions costs can limit the volume of market transactions and, thereby, affect the prospects of a market-based strategy for conservation. In other words, the benefits of the market mechanism for addressing an increasing relative scarcity of genetic resources may, in individual cases, be offset due to substantial transaction costs incurred by the actors in the market.

Transaction costs also occur as a consequence of implementing the market for genetic resources as a policy means of conserving biodiversity. This refers to the costs of developing the institutions that enable market transactions, as well as the costs resulting from changes in the institutional environment and legal system. Figure 9 provides a typology of transaction costs and suggests who is likely to incur the costs, which does not necessarily explain who ultimately bears these costs, since they may be passed onto taxpayers or consumers (McCann et al. 2005).

Regarding the more narrow and traditional focus of transaction cost economics on decision making in the context of (1) market interactions and the

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[83] In focusing upon natural resource management, McCann et al. (2005) provide a good overview of the definitions of transaction costs.

Figure 9:
Transaction Costs Associated with a Market-Based Conservation Policy: A Typology

Type of transaction cost	Incurred by		
	Legislature/courts	Agencies	Stakeholders
Research and information	+	++	+
Enactment and litigation	++	+	++
Design and implementation		++	+
Support and administration		++	+
Contracting		+	++
Monitoring/detection		++	+
Prosecution/enforcement	+	++	+

Notes: ( ) negligible transaction costs; (+) low transaction costs; (++) high transaction costs.

Source: McCann et al. (2005).

(2) structuring of industrial production (Coase 1937; Williamson 1971), transactions with genetic resources are studied from a different perspective: trade in genetic resources is part of a system of contractual relationships in the multistage production process that relies on biotechnological applications.

Regarding this process in general, the literature argues that in order to create appropriate incentives for the sufficient provision of goods and services at each individual stage, property rights should be divided and allocated in a suitable way across the stages (Grossman and Hart 1996). Considering the incentive problems and constraints of the current property right regime on genetic material and embodied information, *contractual relationships* are not likely to have been designed in such a way that an efficient outcome is obtained (Swanson and Göschl 2000).[84]

Accordingly the literature discusses whether transactions in the multistage production process are better organized, on efficiency grounds, within an integrated firm than in a market setting. Considering the industries at stake, such *integration* can occur in two directions: (1) the commercial users of genetic resources, i.e., the R&D-based industries, merge downwards to control the in situ sources of genetic information (*downstream integration*). Alternatively, (2) the providers invest in technology and skilled labor to move closer to the markets for final products (*upstream integration*).

84 Swanson and Goeschl (2000) argue that, given the current property rights regime, it would not be appropriate to apply the Coase theorem to the preservation and exchange of genetic resources because the information as the valuable good is not exclusively connected to the property right to the material that embodies the information.

In practice, the implementation of either of the two integration strategies would include the transfer of property rights across national borders, i.e., either (1) the foreign investment of R&D firms in natural areas or (2) the transfer of technologies to provider countries. Such cross-border transfers are typically associated with high transaction costs, e.g., firms that purchase natural areas in biodiversity-abundant developing countries face transaction costs due to the insecurity of property rights as a foreign landowner. As a consequence, studies conclude that integration cannot be pursued in such a way as to provide an overall efficient (in situ) management of genetic resources as input in R&D (Swanson and Göschl 2000).[85]

Aside from the efficiency criterion, upstream integration is considered a means of obtaining equity in the sharing of benefits from genetic resources. In Section 3.1.2.1, the role of upstream integration is assumed to provide the resource country with a larger portion of the economic rent from genetic information (Artuso 2002; see also Section 3.1.2.1). In this context, the literature questions whether in situ suppliers indeed have a comparative advantage in value added activities with genetic resources. If this is not the case, it is questionable as to whether their investments in value added technologies support the appropriation of the rent by the providers. Consequently, whether upstream integration is an effective means of obtaining equity needs to be assessed carefully (Simpson and Sedjo 1994).

Information Problems

Considering the cycle of a transaction implied by the different types of transaction costs in Figure 9, costs are often represented by the resources that are invested for the gathering of the necessary information in the market environment and potential contract partners. Some information may only be accessible for the economic actors involved at prohibitively high costs. Other information may only be revealed after a period of time. In contrast to this *lack of information*, some information may be available but unevenly distributed among the actors. This problem is known as *asymmetric information*. It can lead to precontractual and/or postcontractual opportunism on either market side and eventually impedes the efficiency of markets for genetic resources and ecological effectiveness with respect to conservation (OECD 2004).

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85 Considering the multistage production process, it is concluded that the current property rights regime on genetic resources virtually does not assign suitable rights to the "best investor" in that process, i.e., the provider in the initial stages. As a consequence, the current management regime lacks efficiency. In other words, a "property rights failure" prevails (Swanson and Göschl 2000; Hart and Moore 1990).

Precontractual opportunism occurs, for example, when commercial users misrepresent the expected or true value of genetic resources in R&D. Also, the providers may ex ante misrepresent the medicinal properties of the material they supply (Laird 1993). Postcontractual opportunistic behavior occurs with regard to the delivery of collected genetic material by the provider or the user's indication of revenues derived from wild genetic information (Simpson and Sedjo 1994). Another variety of ex ante opportunistic behavior is framed as the holdup problem: suppose that the replication of rare genetic information is not possible and, therefore, researchers have an ongoing need for materials from the original source. If, after a time, the provider withdraws from the long-term contract on access to the original source, previous research efforts on the demand side are devaluated. The research-specific costs then represent sunk costs (Samprath 2000).

Several instruments aim at mitigating these information problems. For example, both sides may agree upon an incentive-compatible compensation scheme that includes both guaranteed and contingent payments for the provider (in practice, up-front payments and royalties) (Artuso 1996b).[86] For commercial users, it can be useful to acquire a certificate in order to signal previous compliance with ABS regulations (Glowka 2001). In addition, the fact that some providers have attracted a series of transactions implies that reputation can matter in this respect (Mateo et al. 2001).

Finally, a general lack of information or strong asymmetric information on the market can translate into *diverging perceptions and expectations* concerning the value of genetic resources. In the end, this can cause disagreement among the market participants and impede a settlement between them (Sedjo and Simpson 1995). As a consequence, the market transaction may not take place and the landowners earn little from conservation and the supply of genetic resources.

Intergenerational Externalities

As already mentioned, bilateral contracts on genetic resources may not provide an efficient allocation because of the impact of transaction costs. In addition, inefficiency results from the fact that not all stakeholders can participate in the bargaining. Since *future generations* in particular do not participate, positive intergenerational externalities generated from the preservation of genetic resources are not internalized in present contractual arrangements. Consequently, even in a world with negligible transaction costs and well-defined property rights to wild genetic material, the landowners cannot capture the total value of genetic resources, because of the failure of intertemporal markets (Lerch 1994; von Amsberg 1995).

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86 Since the design of the payment scheme also touches upon the issue of optimal risk sharing, only a second-best solution may be attainable (Samprath 2000).

The Role of Pareto Improvement

In spite of the limitations named, bilateral contracts can at least lead to a Pareto improvement. This is the case when conservation is insufficient in a baseline situation without a contract and the landowner invests the revenues he obtains in additional conservation. In this regard, a positive rent is created (Lerch 1994).

The finding of potential Pareto improvement but remaining inherent sub-optimality draws attention away from the normative question concerning efficiency to the descriptive question concerning the magnitude of the market's contribution to conservation. More precisely, given the specific economic properties of genetic resources, the question is how many natural areas are withheld from conversion for the market supply of these resources. To find an answer in this regard, information on prices and quantities traded on the market is required in the initial step, particularly information on the private value of genetic resources relative to the market values of the goods produced in alternative land use (Frisvold and Cordon 1994). Section 3.3 studies prices and market transaction in more detail.

The Environmental Impact of Bioprospecting

Aside from the limitations that have an influence upon the efficiency of trade in genetic resources, the environmental impact of bioprospecting also needs to be considered. I identify three conditions within the bioprospecting area that ideally are satisfied when the market mechanism is indeed a viable instrument for conservation:

- Natural areas in which bioprospecting is economically profitable represent areas that are rich in biodiversity and simultaneously generate ecosystem services of local and global importance other than genetic resources.
- The extraction of genetic materials for a market supply does not cause any negative long-term externality in the flow of ecosystem services. No ecological thresholds are exceeded. No irreversible ecosystem change is triggered (see sections 2.3.1 and 2.3.3).
- The natural habitat that is rich in biodiversity is threatened by conversion for other commercially more productive uses. This condition is necessary to assess the impact of the commercialization of genetic resources.

In the theoretical studies, as well as in the models introduced in Section 3.2.1, it is implicitly assumed that these conditions are fulfilled.

However, previous bioprospecting has not been sustainable per se. The literature provides evidence that methodologies for on-site selection and collection are often not sensitive to the prevailing ecological conditions and that sometimes the populations of species that are promising for R&D have been largely decimated. Accordingly, these species have become endangered themselves in the end (Dhillion et al. 2002; Oldfield 1984: 132ff.).

Some studies also argue that local communities may actually manage natural habitats in a sustainable way. Under certain circumstances, the increasing demand for genetic resources for commercial use in this regard can dominate traditional use, with potential adverse, long-term effects for the habitats and the resources residing therein (Bhat 1999; Barrett and Lybbert 1999).

In contrast, traditional cultivation in managed ecosystems hosting valuable crop genetic diversity may provide only few other ecosystem services that generate external benefits on a local and global level. The proper management of these ecosystems may support the efficient conservation of crop genetic diversity as an integral part of biodiversity but contribute little to conservation in a wider scope. In these cases, contracts on preserving wild genetic resources in in situ conditions only internalize the externalities from the potential loss of genetic information but not externalities from the decimation of other ecosystem services.

Regarding the threat to converting bioprospecting areas for alternative land use, there is empirical evidence of bioprospecting agreements that have arranged for the right to access genetic resources in protected areas owned by the public sector (Lerch 1994). Examples are supplies by the INBio in Costa Rica or the Yellowstone Diversa contract in the United States. The public sector administration, in this context, controls access to genetic resources, i.e., no open access regime or international common property regime is in place (Reid et al. 1993; Bryson and Kaczmarek 2000). Although protected areas in the public sector do not seem immediately threatened by conversion, the revenues that the public sector obtains from supplying access rights to genetic resources increase the budget for the proper management of protected areas. Consequently, in this case, market returns can also assist conservation in the public sector.

To resume, these caveats imply that the conditions for a substantial positive impact of trade in genetic resources upon biodiversity conservation may not apply in general. However, the hypothesis that the market has a somewhat positive impact upon conservation cannot be dismissed on these grounds. Moreover, assessment of the caveats has to be made on a case-by-case basis. Bearing this in mind, I continue by using the assumption that it has a favorable impact upon conservation. I do so in order to identify further constraints on the impact of commercialization upon conservation. Such constraints may result from an overestimated scarcity of genetic resources relative to other natural resources, as well as input in R&D. The question of relative scarcity constitutes the basis of the analysis in following two sections.

3.3 Trade in Genetic Resources: Empirical Evidence

Markets for genetic resources can only induce conservation incentives if genetic diversity yields a relatively larger return than goods from alternative but biodiversity-poor ecosystem use. In other words, there must be a significant *scarcity* of genetic resources relative to these other goods. When genetic resources for commercial use are exchanged on markets, this scarcity should be expressed in the relative market prices.

Together with observable market turnovers, market prices could be compared to the opportunity costs of biodiversity conservation in order to derive evidence of the potential of a market-based approach to biodiversity conservation. Figures in this regard may explain to what extent markets can contribute to biodiversity conservation, in addition to the regulation of private land use and the public provision of nature protection. In a broader sense, an analysis of these concerns also has to address the question of what the suitable mechanisms for allocating genetic resources are.

A Market Analysis: Fundamental Problems and How to Address Them

An empirical description of the market for genetic resources has to take several *impediments* into account:

- Regarding their commercial use in practice, genetic resources do not usually represent standardized goods whose trade routes can be well traced statistically, as can be done with many industrial goods.
- Genetic resources enter production in several industrial sectors in a differentiated form. Although most of these sectoral productions more or less aim at research, development, and the marketing of new products, the associated multistage production process is arranged quite differently across sectors.
- As far as genetic material enters the R&D process in a processed form, market prices for intermediates in the different stages already contain portions of the value added of inputs other than natural genetic information. Accordingly, the proportional value added genetic resources contribute is difficult to identify.
- Furthermore, in nearly all sectors, multiple economic agents are involved in the exchange and transfer of unimproved and processed genetic materials. Consequently, the genetic resources' route back to their place of origin, as well as the reverse flow of payments associated with them, can hardly be retraced.
- In addition, for certain industrial uses, the biotechnological production of the final product is not based upon the input of original biological raw material in its physical form, but only upon the information regarding its biochemical

structure. This use of derivates adds another difficulty to identifying market values.

- Finally, trade in genetic resources for the purpose of R&D is often kept confidential between the trading partners. In particular, resource buyers seem to be interested in secrecy, since they face ownership risks during the R&D process prior to being vested with an IPR for successful R&D results. Therefore, they seemingly do not want to disclose their purchasing practices with respect to genetic resources to their competitors (ten Kate and Laird 2000).

The combination of these factors impedes systematic and long-term data mining on trade in genetic resources. As a way out, the literature tries to approximate the market for genetic resources by analyzing *market data* for the sectors in which genetic resources are commercially used. Such an analysis is typically structured according to industrial sectors and supplemented by qualitative descriptions of the sectoral R&D processes, with the associated demand for genetic material. This approach also provides a way to classify anecdotal evidence on the trade in genetic resources (e.g., Artuso 2002; ten Kate and Laird 1999; Hill 1999; Swanson and Luxmore 1997).

In order to describe the market in the following, I also use *classification by sectors*. For this purpose, I adopt the seminal study by ten Kate and Laird (1999) as my starting point. In the initial step, I summarize and interpret updated turnover figures for sectors that make use of genetic resources (see Section 3.3.1). Based upon this, I use information in input-output tables to determine the upper limits for the production value of natural resources in pharmaceutical production.

In the second step (see Section 3.3.2), I systematize and analyze data from trade statistics, as well as anecdotal evidence on the exchange of genetic resources, to identify actual market prices and traded quantities. In this respect, I separate the use of genetic information where the information cannot be replicated in ex situ conditions (Section 3.3.2.1) from nonrival use (sections 3.3.2.2 and 3.3.2.3). The results are finally placed into the context of the described market-based strategy for conservation.

In the context of the market description, the issue of national regulations on access and benefit sharing (see Section 3.1.2) is not considered in depth. A detailed analysis of this issue, with its connection to distributional aspects, lies outside the scope of this study. Furthermore, the focus of the analysis falls upon the decisions and actions of private, profit-maximizing economic entities. I only take into account the actions of entities from the public sector if necessary.

3.3.1 Industrial Use of Genetic Resources: A Description by Sectors

Several industries that make use of genetic information can be identified. Existing classifications in this regard refer to sectors (Kate and Laird 1999; Swanson and Luxmore 1997; Hill 1999) or goods that are produced by these industries (Artuso 2002).

Based upon the general description in Section 3.1.1, I distinguish *eight sectors* and subsume them under the *three areas* health care, agriculture, and other biotechnological production. I describe these sectors using annual turnover figures on the world market level.

For some of these sectors, existing studies observe that, although genetic material is used in R&D and production, there are within-sector segments in which *synthetic material* produced in the petrochemical industry completely replaces natural genetic material. This is the case with the development and manufacture of natural personal care and cosmetic products and plant production products. The shares of the sector's segments in which biological resources are indeed used are estimated at 5 percent for personal care and cosmetic production and 2 to 10.5 percent for crop protection (ten Kate and Laird 1999: 188ff., 262ff.).

Regarding pharmaceutical production, there is a long-standing debate on the question as to what extent new pharmaceuticals make use of naturally occurring genetic information. Evidence suggests that many of them are based entirely upon synthetic compounds. Several studies with different foci and applied methodology address this question (e.g., Cragg et al. 1997; Farnsworth and Morris 1976; Grifo et al. to 1996; SAG 1997). Considering the different findings of these studies, ten Kate and Laird (1999, 2000) conclude that 25 percent would be a conservative estimate for the portion of worldwide sales of pharmaceuticals commanded by the products based upon natural genetic resources.[87]

Market Sizes for Genetic-Resource-Based Products

The following figures on market size have, to some extent, the character of a snapshot. Due to significant uncertainty, estimations on future developments are not considered in the description of market sizes below. Table 3 specifies annual turnover figures in billions of dollars. Each figure refers only to the year the data was collected.

Data for the pharmaceutical sector can easily be gathered and is published regularly (e.g., IMS Health 2005; VFA 2004). This is also the case with sales of

[87] For the area of "new" biotechnology, it is assumed that, by definition, genetic material enters the process of R&D and production and could not be replaced by chemical-synthetic substitutes.

Table 3:
Market Size for Goods Based upon Genetic Resources

	Sales on world markets			Products based upon genetic resources	
	Billions of dollars	Year	Data source	Portion of total	Sales on world markets
Health Care					
Pharmaceuticals	400.8	2002	IMS Health (2005)	25%	100.2
Botanical medicine	16.5	1997	Laird (2000)[a]	100%	16.5
Personal care, cosmetics	55.0	1997	ten Kate and Laird (1999)	5%	2.8
Agriculture					
Seeds	30	n.a.	IFS (2004)	10%	30
Crop protection	25.1	2002	ECPA (2004)	2–10.5%	0.5–2.7
Horticulture	16–19	n.a.	ten Kate and Laird (1999)	100%	16–19
Other Biotechnology					
Environmental biotechnologies	57.1–121.1	1998	OECD (1998)	100%	57.1–121.1
Biomaterials, bioenergy	–[b]	1999	ten Kate and Laird (1999)	100%	–[b]

[a]Excluding nutrient supplements. — [b]"Negligible at present, but great potential" (ten Kate and Laird 2000).

crop protection products (ECPA 2004).[88] For seed products, data on commercial sales exists only on a national level. The size of the world market for seeds is estimated based upon data in 49 major producer countries (ASTA 2004; IFS 2004). For horticultural products, data mining is difficult because, on the one hand, different national statistics do not use uniform definitions and classifications for products of agrarian plant breeding. On the other hand, there is strong trade intensity between producers, as well as between producers and intermediaries, which effectively complicates data collection. Against this background, existing figures from official trade statistics are used and roughly adapted (ten Kate and Laird 1999: 158ff.).

Difficulties in classifying and separating related goods also prevail for the products of botanical medicine, which are frequently classified together with nutritional supplements or vitamin and mineral preparations. Figures on turnover in botanical medicine are determined for Western Europe, North America, Japan and China. The size of the world market is estimated upon the basis of this data (Laird 2000). Data on the market for natural personal care and cosmetic products is taken from publications by the Society for Natural Pharmacy and Nutrition Business Journal (ten Kate and Laird 1999: 262ff.).

Figures for new biotechnological production outside pharmaceutical and agriculture production are difficult to estimate. This production can generally be classified according to the above-mentioned functional services that they provide

88 The figures refer to pesticides and do not include transgenic plants.

(see Section 3.1.1) (ten Kate and Laird 1999: 228ff.).[89] However, turnover figures that can be assigned to these services are not available.

In contrast, an important role is often assigned to the *environmental biotechnology* comprising technological applications that help to avoid emissions in industrial processes and/or reduce harmful sediments. Occasionally, the applications for material and energy production are regarded as environmental biotechnologies because they are considered resource-saving compared to conventional technologies. Regarding the four functional subareas specified above, environmental biotechnologies seem to go beyond biotransformations and partly overlap the remaining three subareas (OECD 1998a).

The existing literature frequently does not present market data specifically on environmental biotechnologies but often subsumes such data under environmental technologies in general. The latter includes technologies that rely heavily upon machinery input and human services. While it is evident that biological material is used in the context of these technologies and thereby contributes to the value added, the market value of these technologies is difficult to determine, since most of that value is appropriated in process-integrated biotechnological procedures (OECD 1998a; ten Kate and Laird 1999).

A study by the OECD, therefore, does not consider only sales figures for (1) goods whose development and production results directly from the use of biotechnology (e.g., new environmental services). Figures for goods whose production relies upon (2) improved processes that make use of new biotechnology, and (3) improved processes that make use of biotechnological products from other industrial sectors (e.g., enzymes) should also be assessed. In total, these sales describe the biotechnology related sales (BRS) and illustrate the market penetration of products based upon environmental biotechnologies (OECD 1998a). The major proportion of the BRS-determined market value of environmental biotechnologies occurs in the paper industry (54 to 51 percent), followed by the food and beverage industry (19 to 25 percent), and the pharmaceutical industry (18 to 19 percent). Smaller proportions occur in the chemical industry (5 percent) and the textile industry (1 to 2 percent) (OECD 1998; own calculations).

The market value for the remaining biotechnological production, especially energy generation and the production of materials, in particular plastic, is assumed to be negligible at present but it also points at possible prospects and future market developments in this sector (ten Kate and Laird 1999: 234ff.; *Economist* 2003a). Biomass production for biofuels can be subsumed under agricultural production and is thus addressed by the seed market.

89 Alternatively, "new" biotechnologies may be classified on a product level according to their biological-chemical constitution and the function of the output, for example, according to whether they are enzymes, biocatalysts, or polymerase products (ten Kate and Laird 1999: 228ff.).

Global Sales of Products Based upon Genetic Resources: Findings and Discussion

When adding up the numbers in the table, the annual market size for products based upon genetic resources lies within the range of $220 to $300 billion.

In his study, Artuso (2002) calculates a market size (global sales) of $243.6 billion. In contrast, ten Kate and Laird (1999: 2) calculate a range of $500 to $800 billion, which is due to the fact that they include world market sales for agricultural products up to the final product stage in their calculation.

To discuss ten Kate and Laird's procedure, it is of note that some of the products described represent *goods for final demand*, while others represent *intermediaries* in further production. This is the case in particular with seeds and materials in environmental biotechnology.

Concerning the starting point of the analysis, i.e., identifying the market values of genetic resources and, thus, the potential revenues providers can obtain, the range calculated above includes actual market values together with market values of other inputs, such as capital, labor, and intermediates from other sectors. By adding up the values for processed goods, such as the total of agricultural production, only the value added of other (nonbiological) inputs is included in the information; no additional information concerning the value of unprocessed genetic resources is provided. From this perspective, it is reasonable not to include agricultural produce as a whole.

In the same respect, the figures on environmental biotechnologies presented in Table 3 may be corrected downwards, since, according to the BRS concept, they include the value added of a broad range of goods and services apart from genetic resources. However, in contrast to agricultural production, the figures represent the only reliable information concerning the market value of environmental biotechnologies. Consequently, there is no reliable empirical basis for such a correction.[90]

In this context, I define a general rule as follows: the task is to determine reliable information on the market size for products that are less processed and quite close to the raw material. If this is not possible due to a lack of data, the figures for the goods of the final demand are consulted as the next best option. However, against this background, it is appropriate to consider the figures derived for global sales to be an optimistic upper limit for the value of genetic resources.[91]

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90 This is also the case with horticultural products.

91 Artuso (2002) suggests that the value for biological inputs on world markets amounts to $23.2 billion (excluding the agricultural sector, for which no data were available). However, he does not explain how this figure was derived.

Disregarding the vital role of agricultural production for a moment, which is finally taken into account by the approach of ten Kate and Laird, the market size of the remaining sectors may provide some evidence as to those sectors in which the *market value* of genetic resources is *realized*. Such an assessment finally assumes that genetic material for R&D and manufacture is of similar relevance to each of the sectors, which hardly occurs in practice.

In consideration of this caveat, the data presented here support the following implications:

- The figures in Table 3, combined with information on sectoral production, give the impression that the value added through genetic resources at present occurs primarily in the *pharmaceutical, botanical medicine*, and *plant breeding sectors*, as well in applications of *environmental biotechnologies*.
- The development and manufacture of *crop protection* agents, *horticultural* products, and *natural personal care and cosmetic products* currently seems to be of minor importance to the value added through genetic resources. A relatively smaller market size for the final products in these sectors and/or anecdotal evidence of their relatively strong reliance upon synthetic input as a substitute for natural products supports this conclusion.
- The expectations regarding the strong future market developments that are sometimes asserted for botanical medicine and environmental biotechnology currently lack reliable estimations.
- Regarding seed production, the impact of market-based conservation depends crucially on the extent to which in situ genetic resources in natural habitats or materials that are already stored in ex situ conditions generate market values. I analyze this question in more detail in Section 3.3.2.3.

In the following, I use the figures on market size to provide a more detailed investigation of the industrial production of pharmaceuticals, including botanical medicine.

Drug and Medicine Production in an Input-Output Representation

To reiterate, the contribution of products based upon genetic resources to production is of interest. In the public debate, it is often assumed that the portion of genetic resources in the total production value is relatively small in quantitative terms. In practice, it is hardly possible to determine this portion with sufficient precision. Data from input-output tables for the sectors can provide an approximation in this regard.

I use data on OECD national accounts for three industrialized countries (France, United Kingdom, United States) in order to investigate the composition of the gross production value in the drugs and medicines sector (OECD 2005).

This value is composed of services from 35 other sectors.[92] Since genetic resources do not constitute an individual category in the national accounts, I assume for simplicity that the natural resources indicated in the national accounts are used in pharmaceutical production because of their function as a carrier of genetic information only, i.e., genetic material is represented by the intermediate input of the primary production sectors.

In the national accounts, primary production that processes and transfers natural resources is described by five sectors ("agriculture, forestry and fishing," "mining and quarrying," "food, beverages and tobacco," "textiles, apparel, leather," and "wood products and furniture"). Only agriculture (and, under certain circumstances, the food sector) represents providers of genetic materials.

In Table 4, I determine the share of services from the sector agriculture, forestry and fishing and the gross output in the drugs and medicine sector. Across the three countries selected, this share lies within the range of 0 to 0.5 percent. When the food, beverages, and tobacco sector is generously considered as an additional source of genetic resources and, therefore, its share is added to the agricultural sector, a total share of 1.7 to 4 percent results.

I also assess the extent to which genetic material enters pharmaceutical production via goods produced in other sectors, e.g., intermediates in the chemical industry. Such an assessment is interesting because of the multistage R&D and production process that can be observed in the pharmaceutical industry. For this purpose, the input share of agriculture, forestry, and fishing in each of the 35 sectors is at first computed and then multiplied by the sector-specific share in the production of drugs and medicine. Calculated over all productions, these indirect input portions suggest a share of biological material in pharmaceutical production of 1.2 to 2.1 percent.

Furthermore, the figures illustrate that 4 to 8 percent of the gross output is provided as individual consumption within the sector. Transport, storage, and sales represent 4.3 to 5.3 percent of the gross output value. The value added results from the sum of the payments to employees, indirect taxes, and operating surpluses.

Altogether, these figures give the impression that, at a maximum, 1 to 2 percent of the market proceeds for drugs and medicine can be attributed to natural genetic resources. If the figures on global sales in the health care area derived

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92 The sector corresponds to the sectoral group 3522 according to the ISIC Rev.2. Classification. The definition reads: "the manufacture, fabrication and processing of drugs and medicines, including biological products, such as bacterial and virus vaccines, serums and plasmas; medicinal chemicals and botanical products, such as antibiotics, quinine, strychnine sulfa drugs, opium and derivatives, adrenal, caffeine, codeine derivatives, vitamins; and pharmaceutical preparations for human or veterinary use."

Table 4:
Analysis of Input-Output Tables for the Drugs and Medicines Sector

Sector	Share of gross output (domestic production and imports)		
	United States	France	United Kingdom
Agriculture, forestry, and fishing	0.47%	0.00%	0.01%
Agriculture, forestry, and fishing, together with food, beverages, and tobacco	1.77%	3.95%	1.96%
Agriculture, forestry, and fishing via other sectors	(1.21%)	(2.07%)	(1.39%)
Own consumption	7.5%	4.0%	8.0%
Transport and storage, Wholesale and retail trade	5.3%	4.3%	4.6%
Value added	48.0%	29.0%	52.2%
Compensation of employees	n.a.	20.8%	26.9%
Gross operating surplus	n.a.	6.6%	25.0%
Indirect taxes, net	n.a.	1.6%	0.3%

Source: OECD (2005/OECD National Accounts (1990)),[93] own calculations.

above (Table 3) are generously taken as a basis, the resulting market value of genetic resources in that area would roughly be $2.4 billion.[94] However, this figure relies upon the assumption that the output the agriculture, forestry, and fishing sector has made available exclusively represents genetic resources. As this is unrealistic, the share derived has to be corrected further downwards.

Although $2.4 billion is a considerable amount, it obviously does not represent the revenues that providers of genetic resources can obtain on the market. Due to missing data, however, the actual value of genetic resources in the production of pharmaceutical and botanical medicine cannot be narrowed down any further. Only the anecdotal evidence that I review in Section 3.3.2.2 suggests that the total flow of payments that providers obtain in reality is substantially below the derived upper limit.

In addition, one may ask how much genetic resources and their embodied genetic information contribute to operating surpluses, i.e., the firms' profits

93 Transactions in the input-output table describe domestic and imported transactions at current prices and in national currencies. The data refer to the year 1990. I assume that the proportion of natural resource input in the drug and medicine sector has not changed substantially since then.

94 Artuso (2002) claims that the market value of "biological inputs" in the production of pharmaceuticals and botanical medicine amounts to $22 billion, but he does not explain how these figures are derived.

shown in Table 4. This question concerns the share of the biodiversity surplus in the firms' profits. I speculate that the biodiversity surplus is distributed in favor of the commercial users on the demand side whenever providers are forced to operate at a marginal cost because of competition on the supply side of the market. Against this background, the biodiversity surplus is included completely in the operating surplus net of depreciation.

The figures in the table indicate the portion of operation surplus, including the depreciation of 6 to 25 percent of the gross output. This represents the upper limit for the size of the biodiversity surplus in sectoral production. The actual size of this surplus cannot be determined in this way.

Finally, an analysis using the input-output figure cannot be carried out for the remaining sectors, i.e., plant breeding or environmental biotechnology, since the high degree of sectoral aggregation for the data in the national accounts does not allow any reliable results in this respect.

3.3.2 Transactions with Genetic Resources: Empirical Evidence on Market Values and Quantities

My analysis of the market value of genetic resources has so far pursued a *top down* approach: based upon comprehensive but highly aggregated data for the sectors that make use of genetic material, I have estimated market size. Alternatively, it is conceivable that a *bottom up* approach is pursued where information regarding genetic resources on the level of firms and observations of market transactions are summarized to provide as precise a picture of the market as possible.

In the following, I classify the description of the transactions according to the industrial sectors. In addition, I take into account the different characteristics of genetic resources as economic goods as presented in Section 3.1.1. As mentioned earlier, essentially all sectors focus upon the use of the information embodied in genetic material and the use of the biochemical function associated with the information. For certain sectors, this information can be isolated from the natural raw material and reproduced for all further applications without any further need of the material from the original source. This establishes the property of *nonrivalry*.

For production in other sectors, this *ability to replicate* the information within the individual domain through the use of nonoriginal biological material does not, however, prevail. The desired function that is controlled by genetic information can only be provided if raw material from original sources physically enters the production process, often in necessarily substantial quantities. Genetic resources of this type resemble, in their economic properties, natural resources in conventional production, such as food production.

These two types of genetic resources both have in common the fact that the quality of the goods produced is related to the genetic information embodied. However, their relative *scarcity* is quite differently pronounced, with consequences for the role in R&D and natural resource management.

- For the first type of genetic resources, which I henceforth refer to as *informational use*, the focus falls upon the quest for unknown but valuable biochemical information. Resource management in this regard aims primarily at the preservation of genetically diverse organisms as (yet untested) research options.
- For the second type, which I refer to as natural-resource-like *conventional use*, the relevant biochemical information has often already been identified and the task for resource management is to guarantee a sustainable (in situ) supply of the identified valuable species as a carrier of the relevant information. R&D plays a rather subordinate role and concentrates upon the improvement of the management regime. Under some circumstances, firms invest in R&D efforts in order to overcome the technical barriers to replication, i.e., possibilities to synthesize and rebuild the relevant information are sought.

The differences in the economic properties of the two types cause differences in the size of (market) transactions in order to satisfy the material demands in R&D and production processes, and therefore contribute indirectly to different (market) values for genetic resources. Figure 10 classifies the possible cases when assuming that the screening of a sample of specific material has displayed a promising clue and there is a further demand for material of the same genetic constitution.

Figure 10:
Properties of Genetic Information: Implications with Regard to the Demand for Genetic Material

Genetic information can be reproduced ex situ.	Genetic information cannot be reproduced ex situ.	
Information is isolated and rebuilt on a synthetic basis.	The genetic information is embodied . .	
	. . in (several similar) biological resources hosted at several places.	. . only in a specific species/biological resource hosted only at one specific place.
No ongoing market demand for biological materials from in situ habitats as R&D input.	Positive market demand. Competition among suppliers.	Positive market demand. Suppliers with bargaining power or monopoly suppliers.

Note that the figure conceptualizes the demand after the testing of a sample of one specific biological material by one R&D organization: for uses where genetic information can be reproduced in the domain of the organization, there may only be a substantial market demand under certain circumstances. This depends upon the number of participants in the market who wish to buy just one sample or a set of samples. In contrast, substantial quantitative demand is more likely to occur for uses where genetic information cannot be replicated on a synthetic basis. I develop these aspects in more detail in the following empirical description of sectoral use.

3.3.2.1 Genetic Resources in Conventional Use: Botanical Medicine

The conventional resource use of genetic material, i.e., use where researchers cannot replicate the embodied information in ex situ conditions, is, inter alia, observed for *botanical medicine* and *natural personal care and cosmetics* products. For botanical medicine, plant material is either selectively extracted from natural ecosystems that are relatively undisturbed (wild crafting) or material is cultivated and harvested in modified ecosystems. Commercial intermediaries and retailers often engage in the supply and marketing of the material. In Germany, for example, 1,543 plant species are used as providers of active agents (figures for 1997). According to Lange (1997), use in cosmetics production, as well as in food production and technological applications, is included in this figure. A clear sectoral distinction is hardly possible. 70 to 90 percent of the quantities traded originate from wild habitats, i.e., the material is extracted by wild crafting (Lange 1997: 123f.).

In order to process the raw material, it is extracted, i.e., bioactive substances are separated from the plant material through the use of solvents. The further processing steps involve standardization, which aims to guarantee the uniform quality of the extracts. The products are finally marketed as tinctures or solid natural substances. Anecdotal evidence suggests that R&D in this sector is based less upon capital-intensive laboratory research than on a search for new mixtures of known extracts, as well as on modifications to assure quality (ten Kate and Laird 1999: 78ff.).

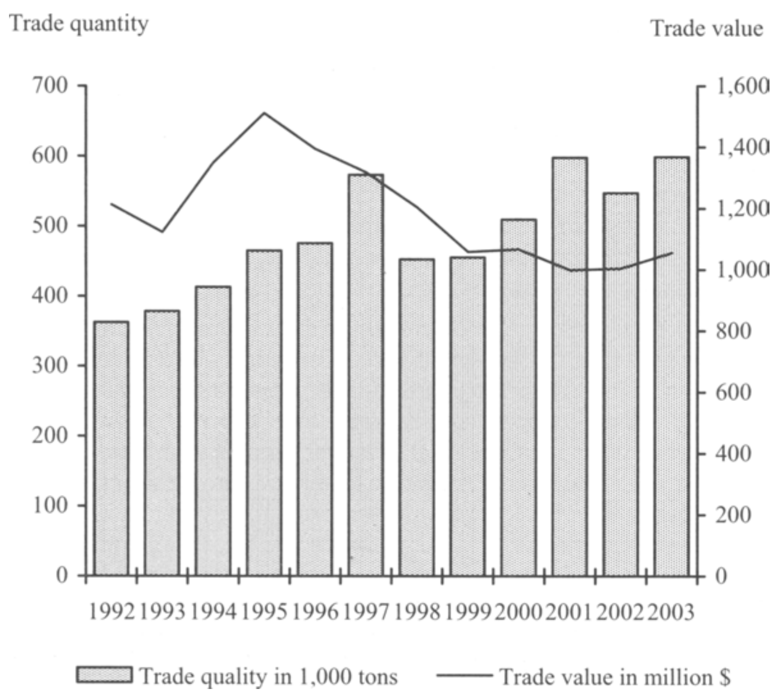
Similar or even identical marketing channels, such as for botanical medicine, are used for natural personal care and cosmetic products. R&D concentrates upon the search for new product components, as well as upon variations of certain existing components (ten Kate and Laird 1999: 274ff.).

Trade with Pharmaceutical Plants: Evidence from Official Statistics

I study transactions with genetic resources as raw plant materials on the basis of official trade statistics. In this respect, the UN Commodity Trade Statistics Database (Comtrade) describes the annual trade flows for "pharmaceutical plants" in its commodity group 2924 (SITC Rev.3). In order to identify worldwide trade quantities and values for these specific genetic resources, I analyze country data for *exports* of pharmaceutical plants. Figure 11 summarizes the data.

The columns in the figure illustrate that, although there are certain fluctuations in the annual data, world trade volume has, in a way, followed an upward trend over the last decade. The data illustrate that, from 1992 to the present, the annual trade volume has increased by approximately two thirds to nearly 600,000 tons per annum. Only in recent years has this increase decelerated.

Figure 11:
Trade Quantities for Genetic Resources as "Pharmaceutical Plants"



Source: United Nations Commodity Trade Statistics Database (Comtrade), own calculations.

However, the rise in trade quantities has been accompanied by a decline in the total trade value. From a peak value of $1,510 million in 1995, it dropped by more than 33 percent to less than $1,000 million in 2001. Since then, total trade value has stabilized at this level. Combining the two variables, shows that the trade value per unit (kilogram) has dropped from $3.35 in 1992 to approximately $1.7 to $1.8 in recent years.

The observed interplay of decreasing prices and increasing market quantities would be consistent with an increasing supply in a perfectly competitive market (where both demand and supply are to some extent price elastic).[95]

Such reasoning relies upon annual average numbers and therefore neglects differences in prices between different plant species supplied by different countries. The data from the Comtrade database shows that within a single year, the value per trade unit in "pharmaceutical plants" can vary significantly between countries. In certain cases, the value indicated for one country exceeds $100 per kilogram or falls below $1. Obviously, these differences between countries are either driven by differences in the costs of collection or value added contained in the traded materials, or are due to resource-specific scarcity.

In order to describe the supply side structure by countries, I calculate the domestic net exports of pharmaceutical plants (including re-exports). For this purpose, I consider only the annual data for 2000, which is the year with the most observations. Figure 12 illustrates that the main providers of these genetic resources are located in Southeast Asia, South America, South Africa, and the Mediterranean area, including Eastern Europe.

In general, these countries are also located in regions known for their plant species richness. Some of these countries also owe their market position to a comparative advantage in the processing of the raw materials. This is apparently the case, for example, with Eastern European countries (Lange 1997: 42). The dominance of the People's Republic of China (including Hong Kong and Macao) implies that the material flows primarily involve products of traditional Chinese medicine, e.g., the materials of the Ginseng root, and that therefore the data mainly represents the area of botanical medicine. The major buyers, i.e., importing countries, are in Western European countries, the USA, Japan, and the Republic of Korea.[96]

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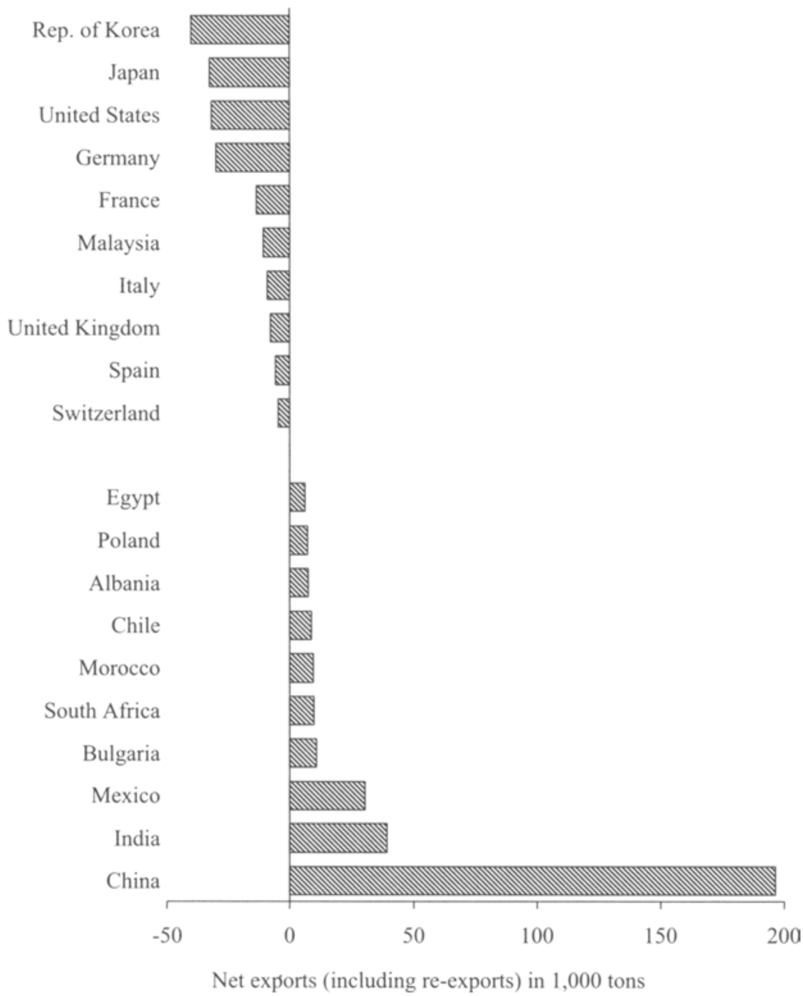
95 A potential reason for this could be that there has been a relatively large profit margin on the supply side, which has attracted new providers, with the consequence that there has recently been a decline in prices. The test of this hypothesis is left to further research.

96 Comparing country data on the value per unit of exported plants, the major net exporters exhibit a relatively low unit value, while major importers exhibit a relatively high unit value.

Figure 12:
Net Exports of Major Exporting and Importing Countries of Pharmaceutical Plants, in 2000



Source: United Nations Commodity Trade Statistics Database (Comtrade), own calculations.

The two figures describe only the *cross-border trade* in genetic resources. *Domestic transactions* are not included in these figures. However, since it is argued that for many countries or regions, top-selling botanical medicine products are often native to the country or region (ten Kate and Laird 1999: 82f.), it may be supposed that the domestic trade in genetic resources in this context is substantially larger than the international trade. This may be true for developing countries in particular, where botanical medicine is often considered to be an important pillar in healthcare (e.g., Cunningham 1997; Le Breton 2001). Unfortunately, data on local markets, especially in developing countries, can hardly be gathered systematically.[97]

Evidence on Prices for Genetic Resources as Raw Materials

Aside from official trade statistics, there is anecdotal evidence on the market price of genetic resources as raw material inputs into the production of botanical medicine. Table 5 summarizes price figures quoted in different case studies. It turns out that the observed prices roughly correspond to the unit values derived from the official trade statistics.

The observed market prices for raw materials, i.e., the revenues an in situ provider obtains, are frequently compared with the revenues earned by suppliers of final products on the market for botanical medicines. In the case study for kava production, it turns out that a kilogram of kava that is processed into botanical medicine has a value of more than \$300 (ten Kate and Laird 1999: 108).[98] GAIA/GRAIN (2000) highlights the fact that for several botanical medicine products, local suppliers of genetic resources capture less than 1 to 13 percent of the sales value of the final product.[99] It is not documented as to which trade quantities correspond to the observed prices, which limits the informative content of these figures.

Profitability and Ecosystem Productivity: A Simple Numerical Simulation

Given the observed unit prices, let us study whether it is possible from an ecological perspective that the promising plants can be extracted from natural areas in sufficient quantities per hectare in order to generate sustainable returns for the landowner. These returns, in effect, must exceed the returns landowners

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[97] For illustration, Cunningham (1993) provides a case study for the Kwazulu-Natal region in the Republic of South African. It turns out that this region already has an annual turnover of 883.8 tons for 39 plant species.

[98] However, in that study, it is not transparent as to how much dry material is necessary to produce extracts of the desired quality.

[99] However, it is not documented how these figures are derived.

Table 5:
Prices for Raw Genetic Material: Figures from Selected Case Studies

Product (species)	Place of origin	Price per kilogram (raw material)	Source
Kava (Piper methysticum)	Oceania	\$5–8	ten Kate and Laird (1999: 108)
Devil's claw (Harpagophytum procumbens)	Namibia	\$0.1–1.8	Cole and du Plessis (2001), Le Breton (2001)
Pygeum africanum (Prunus africana)	Madagascar	\$2	Le Breton (2001)
Rosy Periwinkle (Catharanthus roseus).	Madagascar	\$6	Le Breton (2001)
Gotu Kola (Dentella asiatica)	Madagascar	\$6	Le Breton (2001)
Cat's claw (Uncaria tomentosa)	Peru	\$0.24–0.35	GAIA/GRAIN (2000)
Lapacho (Tabebuia impetiginosa)	Paraguay	\$20	GAIA/GRAIN (2000)

Note: Price per kilogram in Namibia is converted to dollars.

can obtain from alternative uses of land poor in biodiversity. This relationship is illustrated with a simple numerical simulation.

The annual net profits obtained from conservation per hectare, i.e., the kilogram price multiplied by the harvested quantity per hectare, must be larger than the annual net profits from alternative use. When assuming that these alternative profits are constant over time for reasons of simplicity, the net present value, in a way, represents the land price. The other way around, annual net profits can simply be approximated by the product of the observable land price and discount rate. I make the ad hoc assumption that the land price in developing regions rich in biodiversity lies in within the range of \$500 to \$2,000 per hectare and that the discount rate is 10 percent. Thus, the annual net profit from alternative use is \$50 to \$200 per hectare. For conservation to be the more profitable land use choice, the landowner must be able to extract 8 to 67 kilograms of genetic raw material per hectare annually, given a kilogram price of \$3 to \$6. Since labor costs are neglected in this calculation, the actual extracted quantities have to be in the upper area of the interval.

Considering these target quantities, I conclude that certain ecosystems may indeed display the biological productivity needed to provide sufficient material. Consequently, if the extraction costs are relatively low, commercialization can be capable of assisting the effective conservation of natural habitats. However, I can neither qualify for which types of ecosystems or biomes the profitability of conservation is more likely to prevail, nor can I estimate the total area that is conserved in this way.

3.3.2.2 Genetic Resources as Information Goods: Pharmaceuticals

In contrast to its use in the production of botanical medicine and cosmetics, genetic information can be isolated from the natural raw material and reproduced in-house for other uses without the further need for the original material. Such *informational use* of genetic resources is predominant in most of the other sectors, in particular in the plant breeding sector (see Section 3.3.2.3). In this present section, I provide a more detailed description of the pharmaceutical sector and its use of genetic material.

Genetic Resources in Pharmaceutical Production

Genetic resources and their embodied information in pharmaceutical R&D are often referred to as "natural products." They enter the R&D process in various forms: (1) they are used in their pure natural form, (2) they are derived semi-synthetically from the source material, or (3) the informational structure in the source material serves as a model for synthetic products. In addition, genetic material includes (4) molecular compounds, such as protein or polypeptide entities (biopharmaceuticals) that are typically developed by the use of recombinant DNA techniques and reproduced by fermentation (ten Kate and Laird 1999: 40).

Whenever natural products and molecular compounds that are of importance for R&D cannot be reproduced within the domain of the R&D organization, the use of genetic resources as a carrier of information resembles the conventional uses described above. The production of the bioactive compounds Paclitaxel and Conocurvone serve as examples of this (Day and Frivold 1993; OECD 1997: 22).[100] However, in most cases, a (semi)synthesis of the relevant bioactive agents seems to succeed. Therefore, I assume the informational use of genetic resources in pharmaceutical production.

From a biological perspective, the search for promising genetic information in R&D in this sector covers a broad spectrum of resources. In addition to *plant* material, *microbial* material, *animal* material, and *insects* that originate from both terrestrial and marine ecosystems are also screened (ten Kate and Laird 1999: 43f.). Regarding the processing of materials, initially only raw material *samples* are traded. These samples are the source of *extracts* that, in turn, contain *molecular compounds* that are isolated and screened.

[100] In some cases, the technical feasibility of biochemical reproduction occurs over time with ongoing R&D efforts (Day and Frivold 1993).

Extracts of plant material may contain up to 100 different agents (WBGU 2000). A material sample purchased for initial screening displays an average weight of 0.1 to 1 kilogram. 5 to 10 kilograms of dry materials must be processed in order to derive 50 milligrams of pure bioactive compound that suffices for the analysis of the biochemical structure.[101] Until the clinical phases start with an identified agent, a total of 500 milligrams is needed. Generally, up to 100,000 compounds are tested in a R&D project (McChesney 1996; WBGU 2000).

Because genetic information can be replicated, researchers in the pharmaceutical sector establish *in-house collections* of biochemical compounds, either for their own applications or for leasing purposes. According to the data for 1996, these collections, on average, comprise 200,000 compounds. This stock of information is, in effect, increased continuously. Depending upon firm size, 10 to 10,000 new compounds enter into their storage facilities per annum (ten Kate and Laird 1999: 59).

Market Transactions and Bioprospecting Activities

It is of interest how the pharmaceutical industry purchases genetic resources for R&D and what returns the resource suppliers receive. Very little empirical information regarding transactions with genetic resources is available. Moreover, several intermediaries, such as commercial dealers or botanic gardens and university institutes, participate in the transfer from the in situ provider to the commercial user. Only some of the users conduct field collections, but this is obviously not the usual case (ten Kate and Laird 1999: 58).

Empirical observations and case studies provide anecdotal evidence on the relevant market data: in the past, the *informal exchange* of genetic resources was dominant, i.e., genetic material was traded between the individual provider and user like a noninformation good, i.e., a conventional natural resource. Such an exchange was not typically backed by written contracts. Potential claims of the provider on the subsequent use of embodied genetic information were not acknowledged or agreed upon. Moreover, since property rights on genetic resources were sometimes neither specified nor effectively enforced, commercial users were even able to acquire genetic material free of charge.

Data on prices for material samples suggests that in the 1990s, a kilogram of material as dry weight yielded between $50 and $200. For processed materials (extracts), prices ranged between $100 and $250 per sample containing 25 grams

[101] A sample of dry material corresponds to 50 kilograms of plant material or 7–13 kilograms of root material (Laird 1993).

(Laird 1993; ten Kate and Laird 1999: 64). This would correspond to a maximum (gross) market value of $10,000 per kilogram.[102]

Given the previous practices of genetic material exchange, recent political efforts have aimed at reaffirming property rights and strengthening the role of the provider of (in situ) genetic resources. The international regime for *access to genetic resources and benefit sharing* was the outcome of these efforts (see Section 3.1.2). As a result of these institutional changes, transactions with genetic resources and their commercial use are more frequently made transparent in bilateral arrangements on *bioprospecting* (see Section 3.1.3) (Reid et al. 1993).

From case studies on bioprospecting, the following characteristics of such arrangements can be derived:

- Private firms, governmental or partly governmental research institutes or a collaboration of the two act as the buyers on the user side.[103]
- In most cases, the local in situ suppliers and the industrial users are located in different countries, i.e., genetic material is subject to a *cross-border transfer*.
- On a local level, the transferred genetic resources typically are collected either from publicly-owned protected areas, such as *national parks*, or from natural areas (or protected areas) that are *managed by local communities*.
- The supply side in bioprospecting contracts is represented by the agents who manage the biodiverse ecosystems and thus directly control access to the resources. However, *government agencies*, *research institutes*, and *nongovernment organizations* (NGO) are also involved as contracting parties. Their task is to support local stakeholders in supplying genetic resources for industrial use on a national and international level.[104]
- The contractually agreed upon material exchange and compensation is, in practice, often not related to single goods but rather to a *bundle of goods and services*. The services of the supplier may include (besides the transfer of the right to access and use genetic resources) the primary processing of the materials, as well as the transfer of the resource-specific indigenous knowledge. The services of the buyer are typically represented by a combination of monetary and nonmonetary services. Examples of the latter are technology transfer or know-how transfer (training programs) (UNEP/CBD 1998).

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[102] In another study, it is argued that the world market price for synthetic compounds is approximately $1 per milligrams, (thus $1,000 per kilogram). It is also assumed that the price for compounds of natural products is not substantially higher (personal communication in FES (1996)).

[103] A collection of case studies on bioprospecting arrangements can, for example, be found in UNEP/CBD (1998).

[104] Sometimes, the services for specific groups are arranged in individual contracts. As a consequence, the entire bioprospecting agreement represents an extensive network of individual bilateral contracts (Rosenthal 1997).

- Regarding the temporal dimension of bioprospecting activities in one specific place, i.e., the *contract duration* of the bioprospecting arrangements, there is evidence that the individual providers and users in most cases only interact in the short term or medium term. Nevertheless, a provider can contact several buyers simultaneously. For example, InBio, a well-known provider in Costa Rica, has managed to conclude a series of contracts with several commercial users. Mateo et al. (2001) report about a dozen bilateral contracts concluded in the 1990s.

Generally, the specificity of these agreements and heterogeneity of the goods and services exchanged makes it complicated to identify specific market prices for genetic resources.

Information on the *quantities* of genetic material transferred is only available on an anecdotal basis. According to Guerin-McManus et al. (1998), in a project in Surinam, approximately one ton of extracted resources was processed and exchanged. Bonalume and Dickson (cited in Nunes and van den Bergh (2001)) report that a Brazilian company (Extracta) was contracted to deliver 30,000 samples of different biological forms extracted from multiple regions. In most cases, however, the published sections of the bioprospecting contracts do not reveal information on the transferred quantities.

Similar problems exist in assigning monetary values to in-kind compensations, such as technology transfer or training. Also, the monetary compensations are seldom quantified. These compensations have the character of *upfront payments* and/or proportional participations (*royalties*) in revenues or profit generated from marketable products that are developed upon basis of the genetic information contained in the transferred resources. Case studies suggest that upfront payments, at maximum, have reached $0.75 million (CALM AMRAD; OECD 1997) to $1 million (e.g., InBio Merck; Swanson and Luxmore 1997). According to Bonalume and Dickson (cited in Nunes and van den Bergh (2001)), the contract of the Brazilian company Extracta amounts to $3.2 million, whereas no information is given as to how the amount is allocated among up-front payments and royalties.

Concerning royalty payments, there is evidence that the proportional participation in revenues typically lies within the range of 1 to 15 percent. The actual royalty rate corresponds to the degree of processing of the resources the supplier hands over. Empirical studies suggest that for unprocessed raw materials, providers can obtain a rate of 0.5 to 2 percent (ten Kate and Laird 1999: 68).[105] It is

[105] A special agreement between supplier and user has been concluded in Malaysia. Here, it has been shown ex ante that specific genetic information is promising for further development and commercialization. The commercial user, a biotechnology firm, has started a joint venture with the original supplier, a regional government in Malaysia (ten Kate and Laird 1999: 70).

unclear, whether there have been payments based upon a royalty arrangement in any single case.

To resume, the available market prices for material samples in pharmaceutical production lack validity as long as figures on the transferred quantities are missing. Up-front payments and royalties on their own can represent a remarkable amount. However, it remains unclear as to how many providers actually receive such payments. In addition, nonmonetary compensations that are transferred to the providers may display a market value that potentially exceeds the monetary compensations.

The information contained in the accessible bioprospecting contracts neither provides conclusive evidence that trade in genetic resources generates sustainable returns from in situ conservation nor does it imply that the market-based strategy on conservation completely lacks effectiveness. Apparently, a precise assessment can only be made over an extended period of observation and on a case study basis.

This raises the question of the long-term impact of the market upon conservation, particularly in cases where an individual user can reproduce genetic information in his own domain and, therefore, the individual demand for material from the in situ source is relatively low. Durable incentives for conservation in this regard are only generated when providers manage to get paid for the in situ preservation of genetic resources by many commercial users and/or over a longer period of time.

3.3.2.3 Genetic Resources as Information Goods: Products of Agricultural Plant Breeding

The international use of genetic resources in breeding processes in agricultural sector shows features that are different from those of the development and production of crop protection products, which display similarities to pharmaceutical production with respect to the channels for the acquisition of natural products and the arrangements for paying their suppliers (ten Kate and Laird 1999: 188–261).

In the following, I analyze the use in breeding processes in more detail and attempt to condense empirical evidence on transactions and potential market values. The focus falls upon plant genetic resources only, i.e., animal genetic resources are not investigated. Furthermore, it turns out that the use of genetic resources for plant breeding in *horticulture* exhibits certain similarities to their use for the breeding of crops. For this reason, I do not consider this area in the following (ten Kate and Laird 1999: 172ff.).

The Use of PGRFA for Breeding Activities

For the development of crop varieties, breeders search for a new and promising combination of genetic information contained (1) in breeding material that has already been used for other research programs and that is replicated in *ex situ* conditions, and (2) in unexplored materials from *in situ* environments. Generally the search for crop genetic resources with promising genetic information, i.e., traits that confer desirable functional characteristics with regard to productivity or resistance, concentrates upon a narrow range of the same biological species. In a few cases, breeders use material from species that are biologically related. Only recently has the development of new technologies based upon genetic engineering enabled the transfer of genetic information between plants of different species.

Regarding the *biological taxa* of plant genetic resources for food and agriculture (PGRFA) in breeding processes, the breeding and seed production for crops generally only concentrates on very few species: about 30 crop species provide approximately 95 percent of the worldwide food energy supply (FAO 1998: 14).

Since there is a substantial interdependency between regions and countries in the availability of genetic material for breeding programs, crop genetic resources have, historically, been exchanged between the world's different regions (Dutfield 2000a; Kloppenburg and Kleinman 1988). Therefore, the material used in breeding programs can be classified on a country level according to its *geographical origin* as indigenous versus exotic crop genetic resources (FAO 1998: 90).

In addition, the genetic material can be categorized on the level of firms with respect to the degree of human *improvement*. A common classification separates (unimproved) wild species, or, synonymously, wild relatives and landraces, i.e., primitive varieties that result from selection processes by farmers, from modern varieties and breeding lines (IPGRI 1996; Painting et al. 1995).[106]

Crop genetic resources used in breeding programs are summarized as germplasm and presented in different *material forms*. A common classification distinguishes (1) actively growing plants, (2) seeds, and (3) tissue culture. The material is the carrier of the functional units of heredity transmitted to the plant's next generation. Accordingly, conserving the material in this respect aims at preserving the plant's characteristics. As long as this purpose is fulfilled, the material form is of minor importance. Nevertheless, the dominant material form, particularly in ex situ facilities, such as gene banks, is samples of seeds of a particular crop species. These samples are named *accessions*.[107]

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106 Exotic materials are sometimes considered synonymously with unimproved materials (ASSINSEL 1996).

107 For major crops, an accession signifies approximately 500 to 1,000 seeds of the crop species (Virchow 1999b).

While seeds are usually sold on the market as the final products of the *breeding sector*, the suppliers, i.e., the breeders, do not always belong to the private sector. Sometimes, there are collaborations between private and public breeding organizations (Evenson and Golin 1997). Regarding the use of crop genetic resources in *breeding programs*, breeders draw materials of different degrees of human improvement from different sources. According to information from private industry, breeding relies to a large extent upon modern varieties and breeding lines. Landraces, on average, represent only 3 percent of the germplasm used; wild species represent 3.7 percent. These types of rather unimproved material are nevertheless widely used. 94 percent of all breeders asked in the survey reported that they used wild species, whereas 70 percent use landraces (Swanson and Luxmore 1997).[108]

Considering the *course of action*, the breeders usually begin their search for genetic information with germplasm contained in (in-house) working collections. In addition, the search may be expanded to collections held by other breeding organizations, particularly those in the public sector. Only if the search for promising traits has not been successful and if it can be expected that crop genetic resources in in situ environments can make a productive contribution do breeders conduct field collections (Fowler et al. 2000, 2001; Virchow 1999a).

Regarding the *in-house collections*, a survey of the private industry reveals that they are composed to a large extent of their own current breeding programs (on average, 75 percent). Another 15 percent is acquired by exchanging materials with other breeders, 3 percent is obtained from ex situ collections maintained in gene banks, and approximately 1 percent is collected from in situ conservation areas (Swanson and Luxmore 1997).

The World's Gene Pool

The total genetic material available for breeding programs, i.e., including the in situ and ex situ materials of different forms of improvements, is difficult to assess (Fowler et al. 2001). Besides the practical problems in monitoring and identifying it, this is because the stock of crop genetic diversity is subject to *evolutionary processes* in nature and is, therefore, a dynamic variable. Moreover, because of these evolutionary processes leading to the creation of new genetic information, it would be not reasonable to transmit the whole stock of in situ diversity to ex situ facilities (Swanson 1996).

Available ex situ crop genetic resources can be gathered more easily. Figures in this respect refer to the collections of accessions in *gene banks*. At present, the

[108] These figures are basically confirmed by ASSINSEL (1996), stating that materials from an exotic source represent 6.5 percent of the material used in breeding programs. Two thirds thereof originated from ex situ conditions.

number of accessions in 80 crop species is estimated at 6.2 million worldwide. These are stored in approximately 1300 gene banks (FAO 1998: 83; Fowler 2004). Regarding these gene banks, the size of the individual collection varies considerably. The majority of them display a relatively small endowment of approximately 2,000 to 3,000 accessions (Fowler 2004). On a country or continental level, most of these gene banks are located in Europe (496) and Asia (293), as well as in Latin America (227). A smaller number of gene banks are located in Africa (124), North America (101), and the Middle East (67) (FAO 1998: 98). On a supranational level, the genetic resources for major crops are stored in species-specific gene banks hosted at *international agricultural research centers* (IARCs) (Virchow 1999a). The existing 15 IARC combined form the Consultative Group on International Agricultural Research (CGIAR). Their counterparts on a country level are the *national agricultural research centers* (NARCs).
Table 6 classifies the collections of the CGIAR and the NARCs, together with those of the *private sector*, according to the degree of improvement, i.e., shares for the stored material of a different type is calculated for each sector. The figures in the table indicate that gene banks in the different sectors have a different focus. Private sector gene banks, in particular, concentrate upon modern varieties and improved material. In contrast, the CGIAR gene banks place the emphasis upon the preservation of less improved material, i.e., landraces and wild species. In absolute numbers, NARCs store the majority of the 6.2 million accessions (80 to 85 percent) (FAO 1998: 94; Cooper et al. 1994). About 0.66 million accessions are stored and maintained by the CGIAR centers (Koo et al. 2003).[109] The remaining portion is held in private hand (Cooper et al. 1994).

Table 6:
Gene Banks by Sectors: The Composition of the Stored Germplasm

Sector Improvement	Multilateral (CGIAR)	Public sector (NARCs)	Private sector
Wild species	14%	4%	6%
Landraces (incl. advanced landraces)	59%	24%	12%
Advanced cultivars	27%	18%	47%
Others (incl. mixed materials)	–	54%	35%

Source: FAO (1997: 94), own representation.

[109] According to Gollin and Evenson (2003), the CGIAR holds approximately 0.5 million accessions. These constitute 13 percent of all accessions worldwide, including duplicates.

The world's stock of ex situ accessions in decentralized gene banks is subject to change. On the one hand, it can be eroded if unique accessions are inappropriately stored and maintained and are therefore unusable for future breeding programs. This is particularly a threat with accessions stored in gene banks that only have poor facilities and/or a small budget to cover the operational costs (Fowler 2004); on the other hand, the ex situ stock can be expanded when new cultivars or breeding lines are produced and/or unexplored in situ genetic resources are transferred into the ex situ conditions.

Considering the *coverage of unimproved material* in ex situ conditions, recent estimates, which are represented in Table 7, show that for many crops a significant portion of landraces have not yet been collected and stored in gene banks.[110] The figures in the right column of the table indicate that the collection of wild species has often not progressed very far. For some species, significantly less than 50 percent of the landraces have been collected and stored.[111]

Table 7:
Germplasm in Ex Situ Gene Banks: Total Size and Size of Unimproved Material by Crop

Crop	Accessions	Portion of exotic material (wild species; landrace)	Ex situ coverage (wild species; landrace)
Wheat	850,000	n.a.	60% ; 95%
Rice	420,000	1% ; 25%	10% ; 95%
Maize	270,000	<1% ; 17%	15% ; 95%
Sorghum	168,500	<1% ; 18%	10% ; 80%
Millet	90,500	% ; 33%	10% ; 80%[a]
Cassava	28,000	n.a.	5% ; 35%
Potato	31,000	5% ; 12%	40% ; 95%
Sweet potato	32,000	6% ; 16%	1% ; 50%
Banana	10,500	n.a.	n.a.
Common bean	268,500	1% ; 21%	n.a.
Soybean	174,500	1% ; 2%	30% ; 60%
Sugarcane	19,000	n.a.	5% ; 70%

[a]Data only for pearl millet.

Source: FAO (1997: Annex 2).

110 Gollin and Evenson (2003) confirm the figures in Table 7. In their representation, additional information and data on further crops is provided.

111 This finding is at first to be regarded as independent from the normative question concerning whether it would be economically efficient overall to continue with the collection and storage of exotic materials. For this purpose, the (option) values connected with these resources need to be compared to the costs of collection.

Transactions with Ex Situ Crop Genetic Material

Considering the exchange of ex situ materials on an international level, there is empirical evidence of a strong gene flow from CGIAR centers to the public and private sectors in the developing countries in particular but also in developed countries. The majority of the genetic resources belong to improved materials. While the CGIAR facilities distribute approximately 500,000 samples of improved material per annum, the distribution of samples of unimproved materials amounts to only 60,000 to 100,000 samples. Major recipients of the improved materials by the CGIAR are the NARCs in developing countries (72 percent). Furthermore, the CGIAR gene banks also exchange 60,000 samples of unimproved materials between themselves (Cooper et al. 1994; FAO 1998: 281f.; Fowler et al. 2001; Koo et al. 2003; Visser et al. 2000).

In contrast to the CGIAR centers, the transactions of the NARC are not documented in the same detail. Figures in national reports indicate that, in recent years, the inflow or outflow of accessions of unimproved materials in the country has sometimes amounted to 2,000 to 10,000 accessions per annum. Peak suppliers are the national gene banks in the United States, with a maximum of 35,000 to 40,000 samples. Based upon figures for the exchange between national gene banks, certain countries can be identified as net importers (e.g., Brazil, Canada) and others as net exporters (e.g., the United States) of accessions. Nevertheless, in relation to the supranational CGIAR centers, all national gene banks are net importers of accessions. Estimates suggest that a (national) gene bank disseminates on average about 10 percent of its collection per annum (Fowler 2004; FAO 1998: 140, 280ff.).

The gene banks in the different sectors in different countries together form an *informal network* for the exchange of crop genetic material. Recently, the intention has been to put this network on a more formalized basis and foster integration among the international providers of germplasm in the *Multilateral System for Access and Benefit Sharing.* This arrangement is agreed upon in the *IT-PGRFA* (Art. 10–13), although it has not yet been implemented (Fowler et al. 2001) (see Section 3.1.2.3).

Gene banks, particularly the CGIAR gene banks, represent intermediaries between the in situ suppliers of crop genetic material and the users in the breeding industry. Since the intermediaries, as well as many users, have the capacity to replicate genetic resources within their own domain, no further contact with the original supplier and/or the intermediary is necessary.[112] Therefore, the

[112] However, the technical capacities needed for the appropriate storage and reproduction are frequently of insufficient quality. This can be observed particularly for breeders and gene banks in developing countries. Accordingly, identical materials have repeatedly been acquired from the CGIAR and NARCs (Fowler 2004).

transfer between the supplier and the intermediary, as well as the transfer between the intermediary and the user, is not directly related to the actual use of the material.

On the level of breeding activities, germplasm is exchanged between breeding entities, including facilities that are only concerned with (long-term) conservation. Transactions take place (1) between private entities, (2) between the public institutions, and (3) between the private and public sector.

Private firms exchange crop genetic resources either (1) on an informal basis as a *reciprocal material transfer* (frequently without monetary compensation) or (2) by using formal *licensing contracts* (ten Kate and Laird 1999: 145ff.). The latter applies particularly when valuable improved material is transferred. Empirical evidence for the United Kingdom suggests that royalty rates range between 0.5 and 3.0 percent of the income earned by a crop variety and that the total royalty payments amount to approximately $30 million per annum (Swanson and Goeschl 2000). Nevertheless, detailed information on transferred quantities and prices is not reported.

The provision of materials in the public sector, i.e., by gene banks in the NARCs, as well as by the CGIAR centers, to national research institutes and/or commercial breeders has followed the principle of *unrestricted availability*: unimproved and improved materials have been delivered free of charge upon the request of the breeder. In order to control the outflows of material and the delivery costs, sometimes a fee is levied to cover the transport costs. In other cases, individual access is limited in quantitative terms or the lump-sum fee for access to a gene bank's collection is raised (IPGRI 1996; Virchow 1999a). In this regard, the allocation of crop genetic resources is not controlled by the market mechanism. Moreover, genetic resources are administered outside the market. Accordingly, any observable fees cannot be considered market prices.

However, there have recently been efforts to formalize more strongly the gene banks' transfer of germplasm (OECD 1996; Barton and Siebeck 1994). In this context, particularly the NARCs use contractual agreements in the form of *material transfer agreements* (MTAs).[113] These arrangements specify (1) the property rights of the gene bank to possible new varieties that the commercial user develops with the transferred materials, or they specify (2) exclusiveness for the commercial user and regulate the potential repercussions for the gene bank's policy of unrestricted access (see Section 3.1.2.3).

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[113] In order to control the transaction costs associated with such contractual arrangements, the MTAs are frequently standardized.

Transfers from an In Situ Environment: Collecting Crop Genetic Resources

On an international or country level, the international gene banks, i.e., the CGIAR centers, play a major role in the ex situ storage and distribution of unimproved crop genetic resources, namely wild genetic resources and landraces. Empirical evidence suggests that there were major collection activities in the mid-1970 and mid-1980s. From the peak figures in the annual entries of about 5,000 accessions, the inflow has decreased to about 2,000 accessions per annum over the last decade. This deceleration is attributed to the fact that the capacity to conserve germplasm from in situ sources has improved on a national level. Accordingly, the collection and preservation of germplasm is increasingly being organized on a country level (FAO 1998: 86f.; Fowler et al. 2000).

There are also cases where the exchange of material from in situ sources is arranged in a bilateral contractual arrangement between countries or, more precisely, the public sector research institutes of different countries (Brazil-Malaysia Agreement on Hevea; IPGRI (1996)). However, explicit market prices for genetic resources cannot be identified in this connection.

Crop genetic resources transferred from in situ environments frequently involve material from the *centers of origin (and diversity)*, i.e., the areas and places where the plant species is historically endemic and/or was first domesticated. The centers of origin for major crops are predominately, but not exclusively, located in the developing world. Areas in these countries frequently constitute semi-arid or mountain ecosystems with low agricultural productivity and resource-poor farmers who often manage them (FAO 1998: 20ff., 501ff.; Kloppenburg and Kleinman 1988).

On an individual basis, the use and, thereby, the in situ preservation of crop genetic resources in these areas rich in genetic diversity is often organized in *informal networks* of small-scale farmers who exchange seeds on a local level (Bellon 2004; Brush 2002). The transfer of resources from these networks to the outside world usually takes place in the form of an informal transfer of the material. For example, the breeder or intermediary acquires a sample of a plant or fruit on the local market, where the supplier is only paid the material value. Further property rights or claims to payments for the use of the material's informational content are not considered (ten Kate and Laird 1999: 132,155).

While in situ preservation and the use of landraces can take place in a private farming environment (*on-farm conservation*, FAO 1998: 59ff.), the in situ management of wild relatives is often organized as joint product of environmental protection, as is manifested in the establishment of *protected areas* (FAO 1998: 54ff.). Since protected areas are mostly controlled by the public sector, market forces therefore do not drive in situ conservation. Chapter 4 considers the economics of protected areas in more detail.

In contrast, when regarding private farming and potential compensation payments for crop genetic resources with reference to the *farmers' rights* principle (see Section 3.1.2.3), there have been conceptual considerations about the implementation of a multilateral trust fund that could finance such compensations. Current discussions are considering the question as to how contributions to the fund could be determined. The focus in this regard falls upon royalty payments on a country level that should somehow reflect the received inflow of crop genetic resources.[114]

Besides this discussion, a Global Crop Diversity Trust has been established under the roof of the FAO in 2004. This fund will primarily raise and invest funds for the maintenance of gene banks, as well as the preservation of crop genetic diversity. The fund is to be endowed with $260 million, which is to be contributed both by governments and the private sector. It is, however, argued that the fund should not be regarded as the formal mechanism for benefit sharing called for in the IT-PGRFA (Hawtin 2004).

To summarize, the international allocation of crop genetic resources is only partially controlled by the market mechanism. This is particularly to due the public sector participation on the supply side of genetic resources, as well as because of the somehow nonexclusive access to unimproved and improved germplasm in ex situ conditions. Consequently, incentives for the in situ conservation of biodiversity are limited due to the inexpensive supply of ex situ material.

3.4 Trade in Genetic Resources from a Theoretical Perspective

Aside from descriptive analyses of the market for genetic resources, another class of studies uses theoretical models to determine the private value of in situ genetic material in R&D. Some of these studies also make use of numerical simulations. They focus upon how much firms using genetic resources in R&D are willing to pay for material samples and how much (in situ) providers of genetic material can earn on the market.

Given the varying character of genetic resources in the various sectors, existing studies concentrate upon specific sectoral uses of genetic resources, particularly those of the *pharmaceutical sector*, and try to determine *implicit prices*. For this purpose, the studies use observable data related to prices, such as

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[114] According to a simulation for payments for the genetic variety of the common bean, it turns out that the payments might be relatively modest in size and the net flows of payments are predominately from those developing countries with a small basis of genetic variety to other developing countries with a larger basis (Pachico 2001).

revenues in markets for final products (see Section 3.3.1), empirical estimates for success probabilities in R&D, or scientific knowledge on the diversity of the biological species in a given natural area.

The study that has attracted much attention and stimulated subsequent research (Rausser and Small 1998, 2000; Craft and Simpson 2000; Costello and Ward 2003) is the study by *Simpson, Sedjo, and Reid* (1996), whom I will refer to as SSR henceforth. The partial equilibrium model introduced in this study is in the focus of my analysis.

In the following section, I introduce the SSR model and the related literature. I also examine the parameterization of the SSR model in the numerical simulation, since several studies meanwhile provide more recent data on biological and economic parameters used in the model.

In Section 3.4.2, I summarize the findings of analytic studies that represent the characteristics of pharmaceutical R&D differently than SSR do and discuss the resulting impact upon the value of genetic resources as it is reflected in the SSR model. In Section 3.4.3, I study the influence of alternative ecological conditions upon private value. I initially stay within the SSR framework and consider different assumptions with regard to the bioprospecting environment. Then, I introduce the approach for valuing genetic resources proposed by Goeschl and Swanson (2002a, 2003a, 2003b) and discuss its relationship to the SSR model. In Section 3.4.4, I finally review analytic studies on the potential private value of genetic resources in the agricultural sector and assess the literature with respect to the described strategy of conservation by commercialization.

3.4.1 The Private Value of Genetic Resources in Pharmaceutical R&D: The Concept of Redundancy

In introducing the SSR approach, it is helpful to remember the major modeling approach that preceded the SSR model: this is the approach of the *average value* of genetic resources that is applied in studies by Farnsworth and Soejarto (1985), Principe (1989), Pearce and Puroshothaman (1992), Aylward (1993), Mendelsohn and Balick (1995), Barbier and Aylward (1996), and Artuso (1996a).[115]

The average private value of genetic resources is roughly derived as follows. When considering the collection of (plant) species that come into question for the search for promising genetic information, the value of that collection is assumed to be equivalent to the expected net value of a newly developed

[115] This literature is reviewed, for instance, in OECD (1997, 1998b) and Cartier and Ruitenbeek (1999).

pharmaceutical. Accordingly, the willingness to pay for an individual species included in the collection is equal to the expected total value of the entire collection multiplied by the probability that a previously untested species will be successful in R&D (Cartier and Ruitenbeek 1999). This value is finally interpreted as the *willingness to pay* on the part of the commercial users of genetic material.

To illustrate the general modeling approach in analytic terms, let us consider Pearce and Puroshothaman's study (1992). The value of specific land, A, hosting genetic resources is given by the linear equation

$$(3.24) \quad V_A = p_A V_{ph} \kappa, \text{ with } 0 < p_A < 1,\ 0 < \kappa < 1.$$

The value of a newly developed pharmaceutical is V_{ph}, where p_A denotes the probability that the collection of genetic resources hosted in area A contains the genetic information that yields the new product. The parameter κ summarizes data on the relative scarcity of the R&D input other than natural products, as well as information on the allocation of bargaining power between the providers and users of genetic resources.[116]

In contrast to this average value approach, Simpson et al. (1996) apply the *value of a marginal resource* when describing the users' willingness to pay. The intuitive reason for this approach is that under the assumption of perfect competition, factors are generally paid their marginal product.

The Value of the Marginal Species: The SSR Model

To derive the value of the marginal resource, the R&D process is modeled upon the basis of search theory: the search for useful genetic information is represented by the sequential testing of samples of genetic material for a desirable biochemical property that is distributed stochastically over a given collection of samples (Evenson and Kislev 1976; Weitzman 1979).

To determine the value of the collection to the researcher, only the unit costs of testing, c, the revenues, R, in case of the successful development of a new product, and the probability, p, that a single genetic resource will lead to the new development have to be specified.

The expected net revenues from testing a single genetic resource are $pR - c$. If the tested resource indeed contains information that is suitable for the development of a new product, the search process is terminated. This is because it is assumed that when a discovery has been made, all remaining untested resources only contain redundant information. However, with a probability of $1 - p$, a

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116 Pearce and Puroshothaman (1992) use different notations and interpret the variable κ in a more differentiated way. For convenience, I choose a simple representation of the model without changing the model's basic features.

tested resource will not display any promising genetic information. In this case, it is removed from the collection of research options and the search continues by testing the next resource and so on. This representation of the R&D process treats each resource as an independent Bernoulli trial with an equal probability of success. Given a collection of n genetic resources, the value of the collection is

$$(3.25)\quad V(n) = pR - c + (1-p)(pR-c) + (1-p)^2(pR-c) + .. + (1-p)^{n-1}(pR-c)$$
$$= \frac{pR-c}{p}\left(1-(1-p)^n\right).$$

Following these conventions, the value of the nth genetic resource is $v(n)$: [117]

$$(3.26)\quad v(n) \equiv V(n+1) - V(n) = (pR-c)(1-p)^n.$$

This value of a marginal genetic resource strongly depends upon the assumption of *redundancy* among genetic information contained in different genetic resources. According to Simpson et al. (1996), redundancy occurs because

- specific genetic information can be contained in all individuals of the same species or
- the same genetic information may be found in different species or
- different genetic information can provide the same therapeutic mechanism.

In all of these cases, those individuals of a species with the relevant information in excess of the number of individuals necessary to preserve a viable species' population are considered redundant.

The redundancy assumption influences the resource value in two ways: the value of a marginal resource, $v(n)$, is smaller,

- the larger the collection of genetic resources, n, considered for testing, simply because of the declining relative scarcity of genetic resources, or
- the higher the probability, p, that it contains promising information. This is because in the sequential testing of many resources, the probability of a discovery being made before a specific resource is tested increases with an increasing p. In the event of a discovery, the search is terminated and a zero value is assigned to all untested resources. This, in turn, influences the expected value of marginal resource.

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117 Equations (3.25) and (3.26) correspond to equations (1) and (2) in Simpson et al. (1996).

Numerical Simulation with the SSR Model

The model as it is presented in equation (3.26) is used in a numerical simulation in order to derive numerical results for the resource value. In this context, Simpson et al. (1996) specify the parameter values of the model in the following way: the considered collection of genetic resources described by n is defined as the set of all plant species existing on earth, which is denoted by N. It is assumed that this collection is used in a specific number of independent research projects per annum. To model the value of a marginal resource contained in N in a sequence of projects, a scaling factor, λ, is introduced into equation (3.26). The resulting expected annual payoff of a single genetic resource is finally divided by the discount rate, δ, to obtain the resource's net present value:

$$(3.27) \quad v(N) = \frac{\lambda}{\delta}(pR - c)(1-p)^N .$$

As mentioned earlier, success probability plays a crucial role: for the numerical simulation, Simpson et al. set p equal to a value that maximizes the value of a marginal resource in order to derive the upper limit of the R&D firms' willingness to pay.[118] They calibrate the unit cost, c, on the basis of the observed out-of-pocket cost for the development of a new drug, K, and the assumed number of genetic resources in the collection considered for R&D, i.e., $n = N$. The expected revenues, R, are calculated by assuming a revenue-to-cost ratio of 1.5. The value for λ is set equal to the average number of annual drug approvals, q, observed, conditional upon the probability that the sequential testing of the entire collection yields a new development.[119]

The net present value is further used to calculate the firms' willingness to pay for the preservation of natural areas as habitats of genetic resources, or, synonymously, species. In order to describe the connection between the size of the natural areas and the richness of promising species hosted therein, Simpson et al. use a functional form of species-area relationship based upon the theory of island biogeography in ecology. The first derivative of this species-area relationship describes the change in the observed species richness from a given change in the size of the preserved natural area. Formally, this first derivative represents

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118 There exists a unique success probability that maximizes the value of a marginal species, since the marginal product depending upon the success probability increases at first but then decreases due to the declining relative scarcity of genetic resources.

119 Given a collection of N species and an average number of q drug approvals with a constant share of η for newly approved drugs based upon natural products, the scaling factor is defined as $\lambda = \eta q / 1 - (1-p)^N$, with $q > 0$ and $0 < \eta < 1$. I have introduced q and η to describe the calculation more effectively. These variables are not explicitly denoted in the original paper.

the product of the species density, D, i.e., species per hectare and the species-area elasticity, denoted by z. Multiplying the net present value of marginal resource by these two parameters yields the value of a marginal hectare of natural habitat.

For the numerical simulation, Simpson et al. take the empirical data for 18 *biodiversity hotspots* described by the absolute number of species and the hotspot-specific species density, D_i. The elasticity for species richness with respect to the preserved area, z, is assumed to be 0.25, which is a widely used parameter value in the ecological literature. The value of the marginal hectare in the numerical simulation is defined as

$$(3.28) \quad \frac{\lambda}{\delta}(pR-c)(1-p)^N zD_i, \ \mathrm{i}=1..18.$$

D_i is the observed number of species per unit of area in the hotspot i.

Land values for the hotspots calculated in this way range from \$20.63 per hectare to less than \$1. These results are interpreted in such a way that the R&D industry's willingness to pay for the preservation of situ environments of genetic resources is insufficient even in biodiversity hotspots to compensate for the private costs of conservation. Without being explicit about these costs of conservation, it is implicitly assumed that income generated from land use other than conservation is greater than the derived figures on attainable prices for conservation. Consequently, the trade in genetic resources could not guarantee the preservation of genetic diversity and could contribute little to biodiversity conservation as a whole.

Responses to the SSR Model

The most prominent reaction to the SSR model and its numerical results is Rausser and Small's study (2000). In their model, the same setting for the search process is applied as in Simpson et al. (1996), except that resources of a different genetic constitution are not perceived to be homogenous but rather to be *heterogeneous inputs* in the R&D process. More specifically, it is assumed that because of publicly available information, the researchers are able to discriminate between promising and less promising genetic material in advance of the search. In this regard, they assign differentiated success probabilities to the different materials. After ordering samples of genetic resources with respect to the observable probabilities, p_n, for $n=1..N$, the sequential testing starts with the most promising ones.

As a consequence of the additional information, the expected search costs can be reduced substantially compared to a sequential search with a random order of samples assuming the SSR model. Prior information assists the creation of "information rents" that lead to a comparatively higher "incremental value" of a genetic resource and, therefore, to a higher willingness to pay for preserving the natural habitats of these resources.

Rausser and Small (2000) use genetic resources and species as synonyms and define a value function, $V_n = V_n(R, p_1..p_N, c)$, to express the incremental value of the nth species, denoted by v_n. V_n represents the expected value of continuing the search with the nth species after testing first, $n-1$, species has not been successful. The incremental value, $\hat{v}_n$, presents an R&D firm's maximum willingness to pay for a "call option" on the nth species at the beginning of the search. It is defined as the difference between the expected net payoff at the outset of the search, i.e., V_1, and the expected value of continuing the search in a case where the nth species does not display any useful information—much like the $(n-1)$ species that would have been tested before, i.e., V_{-n}.[120]

In order to study how information rents influence the value of marginal species compared to the SSR model, the authors provide a numerical simulation with the same empirical data on the parameters R and p, as well as on the biodiversity hotspots used in Simpson et al..[121] They discover that the industry's willingness to pay for land in the hotspots lies between $9,177 and less than $1 per hectare. The authors argue that these results imply that bioprospecting can indeed create significant incentives for biodiversity conservation. Thus, their results challenge the findings of Simpson et al.

In addition, the authors use empirical evidence on R&D processes to explain why they believe that their modeling approach represents the actual R&D activities more effectively. Their main argument is that, in reality, the search is conducted in a more *directed way* than in a *random way*, as the SSR model implies. It is implied that because of the assumption of a random search, the figures derived by Simpson et al. systematically understate the value of genetic resources.

In response to the two studies, Costello and Ward (2003) question whether only the assumption of a more directed search accounts for the differences in the derived numerical results. They recalculate the incremental value in Rausser and Small's model by deliberately choosing search orders that strongly deviate from the order of descending success probabilities. It turns out that the incremental value calculated with an arbitrary order is indeed below that with a correct order of descending probabilities, but only to a relatively small extent. Costello and Ward therefore conclude that the prior information on success probabilities and

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[120] For the analytical representation of the value function and the incremental value, see the equations (1) to (5) in Rausser and Small (2000).

[121] Compared to the SSR model, the value for the unit cost, c, is adjusted on an ad hoc basis.

the associated information rent cannot be the major reason for substantial differences between the Rausser and Small and the Simpson et al. results.

Based upon a *comparison of the numerical simulations* contained in the two studies, it turns out that the differences in the derived numerical values of genetic resources are only partly driven by the assumption of heterogeneity. To a large extent, the differences result from differences in the numerical values assigned to parameters other than revenues, R, and success probability, p, such as, for example, coefficients in the species-area relationship. From this finding, Costello and Ward conclude that, without favoring one of the models, a further judgment of private incentives to preserve genetic resources crucially depends upon "empirically credible parameter values" and a "sound defense" of the resulting estimates.

Numerical Simulations with the SSR Model: An Empirical Parameter Update

Against the background of Costello and Ward's discussion, I update the original numerical simulation provided in Simpson et al. by using more recent data. First, I consider alternative values for the economic parameters only. In a second step, I introduce new numerical values for the ecological parameters.

The per-hectare values of the hotspots are determined by the use of equation (3.28). It is shown that because of a constant value for a marginal species, the differences in land values among the hotspots are solely driven by differences in observable species density, D_i. When new data regarding the economic parameters are used, this does not change the order of hotspots with regard to the land value, i.e., the highest value is derived for the hotspot with the largest species density. Therefore, I only study the maximum values per hectare in the following.

To illustrate the sensitivity of the SSR model, I update the *economic data* in sequential steps. At first, I only introduce new data for the number of new products and their development costs (the previous and updated parameter values are presented in Table 8).

- The expected number of new products, q: based upon figures by CMR International cited in VFA (2003), on average, 37.1 new molecular entities (NMEs) were launched on world markets between 1992 and 2002.[122]
- The cost of developing a new product, K: Di Masi et al. (2003) have updated their figures on the estimates from an earlier study (Di Masi et al. 1991), which were used in the original simulation. Estimated out-of-pocket costs per

[122] In contrast, Simpson et al. only use data on approved drug applications in the United States. Furthermore, they employ cost estimates by Di Masi et al. (1991) that relate to the development cost of "new chemical entities (NCEs)." NMEs include both NCEs and new biological entities (NBEs).

Table 8:
Changes in the Value of a Hectare of Bioprospecting Habitat: Summary of Results from an Updated Numerical Simulation

Parameter	Original value	Updated value	Impact on land value[a]
Expected number of new products	23.8	37.1	↑
Cost of developing a new product	\$300 million	\$802 million	↑
Discount rate	10%	11%	↓
Share of drugs derived from higher plants	33.3%	25%	↓
Revenue-to-cost ratio	1.5	1.28	↓
Alternative hotspot data	18 hotspots	25 hotspots	↑
Plant species richness	250,000	300,000	↓
Species-area elasticity	0.25	0.3 (ad hoc)	(↑)

[a]Changes in land values relative to findings in the SSR simulation.

new drug are capitalized to the point of marketing approval at a real discount rate of 11 percent. By this method, which is similar to that in Di Masi et al. (1991), the total preapproval costs are estimated to be \$802 million (2,000 dollars).[123]

Using these new figures, the values of other parameters also change: by maintaining a revenue-to-cost ratio of 1.5, R rises to \$1,203 billion. Since the screening cost per unit and the maximizing success probability are interdependent, their values have to be adjusted in an iterative way.[124] Due to the rise in the total cost, the unit cost, c, is \$10,876. Accordingly, p is 13.04×10^{-6}. The probability of a success within the entire collection increases to 96.16 percent.

Based upon the data set updated for the two parameters, the willingness to pay per hectare of bioprospecting area for the hotspot with the highest species density (Western Ecuador) is \$51.95, which is 152 percent higher than the value in Simpson et al.'s original simulation.

This increase in the maximum willingness to pay can be attributed to three factors: (1) the broadening of the scope by considering new drugs worldwide, (2) the increase in expected revenues that is driven by the increase of development

[123] The cost estimate finally determines the monetary dimension of the numerical results. In their 1991 study, Di Masi et al. derive an average cost estimate of \$231 million (1987 dollars) by capitalizing out-of-pocket costs to the point of marketing approval at a 9 percent discount rate. Simpson et al. round this figure up to \$300 million.

[124] The values for c and p must simultaneously satisfy the following equations taken from Simpson et al.: $p = R + nc/(n+1)R$ and $c/p = K/1-(1-p)^n$, where K is the total cost of R&D.

costs in real terms and the assumption that the ratio of revenue to cost remains the same, and (3) the increase in the overall price level.

In the following, I modify the numerical values of additional economic parameters and study how the maximum value of $51.95 changes when the described new data for the number of drug approvals and development costs are used and only one additional parameter is adjusted.

- The discount rate, r: For reasons of consistency, the discount rate can be set at 11 percent (as in Di Masi et al. 2003) instead of 10 percent. This 1 percent increase would yield a maximum willingness to pay of $47.23.
- The proportion of drugs derived from higher plants, η: As already mentioned in Section 3.3.1, it is controversial as to what extent new pharmaceuticals make use of naturally occurring genetic information or how many of them are completely based upon synthetic compounds. According to the findings by ten Kate and Laird (2000), a proportion of 25 percent of all new drugs presents a "conservative estimate" for pharmaceuticals based upon natural products. Simpson et al., in comparison, assume a proportion of one third. By using a parameter of 25 percent instead of 33 percent, the maximum willingness to pay per hectare is estimated to be $38.96.
- The revenue-to-cost ratio, *R*/*K*: Simpson et al. use a "generous estimate" of 1.5:1 that is calculated on the basis of reported data on R&D expenditures, costs, and revenues. The ratio can be related to figures on the net profit margin, which are often used to describe the economic performance of the industry. A simplified representation of this margin is $1-K/R$. Consequently, the assumed ratio corresponds to a margin of 33.3 percent. In contrast, a recent description of the pharmaceutical industry by Reuters (2005) identifies a net profit margin of 22.3 percent, which translates into a revenue-to-cost ratio of 1.28:1. When using this ratio in the model, the maximum willingness to pay per hectare falls to $10.89, which is substantially below the value derived using the original data in Simpson et al. (1996). Nevertheless, even at this level of revenue to cost, the expected value of conducting the screening of any genetic resources is still positive, i.e., $pR-c>0$.

Table 8 summarizes the changes in the value of a marginal hectare in a biodiversity hotspot when using new data input in the SSR simulation. To resume, the numerical results based upon the new data for parameters do not deviate significantly from the original results presented in Simpson et al. The maximum willingness to pay for a hectare of bioprospecting area ranges from $10.89 to $51.95 for sets of parameter values that have been updated by new data in various combinations.

In addition to new data on the economic parameters, new *ecological data* has also been generated. The hotspot study underlying the original numerical simulation is updated and broadened by Myers et al. (2000): the 25 (previously 18) hotspots display a total area of 212 million (75 million) hectares, which, in total, host 133,149 (39,410) endemic plant species.[125]

- New hotspot data can be applied to calculate land values. I again only use new data on the expected number of new developments, q, and the development cost, K, and, otherwise, the parameter values in Simpson et al. The willingness to pay for the hotspot with the highest species density, which is the Eastern Arc Mountains and Coastal Forests of Tanzania and Kenya, is \$44.60 per hectare.
- Species richness among plants, N, is reassessed. Myers et al. (2000) argue that the reliable number of plant species worldwide is 300,000 (previously 250,000), which expands the collection of genetic resources considered in the simulation by 20 percent. Accordingly, the success probability, p, is reduced to 10.87×10^{-6}. The estimate for the maximum willingness to pay is \$37.08 per hectare.
- Species-area elasticity is typically represented by $z = 0.25$, which is a widely agreed upon figure. Empirical estimates of the elasticity in ecology generally fall within a narrow range (Armsworth et al. 2004). In this respect, let us assume some reasonable values for which land values are upwardly biased, e.g., $z = 0.3$. Simulation results show that the maximum willingness to pay rises to \$44.50 per hectare.

To resume, for the *updated hotspot classification*, the derived maximum values are \$37.08 and \$44.60 per hectare. Table 9 provides a complete description of the updated hotspots. The choice and order of variables depicted in the table is identical to that in Simpson et al. The hotspots are ordered according to their value per hectare. This value is driven by the endemic plant species diversity. The column on the right displays the values per hectare when the three parameters, i.e., q, K, and N, are updated. The last column describes the resulting land values when the changes in the other three parameters r, η, and R/K are taken into account.

[125] The figures in brackets indicate the data in a previous study by Myers as reported in Simpson et al.

Table 9:
Maximum Willingness to Pay to Preserve a Hectare of Land in Biodiversity Hotspots

Hotspots	Remaining primary vegetation (1,000 hectares)	Number of plant species	Proportion of species endemic to region	Endemic plant species per hectare	Maximum willingness to pay (empirical update for 3/6 parameters)	
Coastal Forests of Tanzania/ Kenya[a]	200	4,000	0.38	0.00750	$37.08	$5.28
Philippines	902	7,620	0.77	0.00646	$31.96	$4.55
New Caledonia	520	3,332	0.77	0.00491	$24.26	$3.45
Polynesia/Micronesia	1,002	6,557	0.51	0.00333	$16.44	$2.34
Cape Floristic Province (South Africa)	1,800	8,200	0.69	0.00316	$15.61	$2.22
Caribbean	2,984	12,000	0.58	0.00235	$11.60	$1.65
Western Ghats/Sri Lanka	1,245	4,780	0.46	0.00175	$8.66	$1.23
Madagascar[b]	5,904	12,000	0.81	0.00164	$8.13	$1.16
Southwest Australia	3,334	5,469	0.79	0.00130	$6.42	$0.91
Sundaland (Indonesia)	12,500	25,000	0.60	0.00120	$5.93	$0.84
Mediterranean Basin	11,000	25,000	0.52	0.00118	$5.84	$0.83
Brazil's Atlantic Forest	9,193	20,000	0.40	0.00087	$4.30	$0.61
Indo-Burma	10,000	13,500	0.52	0.00070	$3.46	$0.49
Succulent Karoo (Southwest Africa)	3,000	4,849	0.40	0.00065	$3.20	$0.45
Tropical Andes	31,450	45,000	0.44	0.00064	$3.14	$0.45
South-Central China	6,400	12,000	0.29	0.00055	$2.70	$0.38
Choco/Darien/Western Ecuador	6,300	9,000	0.25	0.00036	$1.77	$0.25
Caucasus	5,000	6,300	0.25	0.00032	$1.58	$0.23
New Zealand	5,940	2,300	0.81	0.00031	$1.55	$0.22
Wallacea (Indonesia)	5,202	10,000	0.15	0.00029	$1.43	$0.20
California Floristic Province	8,000	4,426	0.48	0.00027	$1.31	$0.19
Mesoamerica	23,100	24,000	0.21	0.00022	$1.07	$0.15
Central Chile	9,000	3,429	0.47	0.00018	$0.88	$0.13
Western African Forests	12,650	9,000	0.25	0.00018	$0.88	$0.13
Brazil's Cerrado	35,663	10,000	0.44	0.00012	$0.61	$0.09

[a]Including Eastern Arc. — [b]Including Mauritius, Reunion, Seychelles, and Comoros.

Source: Myers et al. (2000), own calculations.

3.4.2 Alternative R&D Settings and Their Impact upon Private Resource Value

The updating of the empirical input in the numerical simulation illustrates that the results of the SSR model seem to be relatively robust to changes in parameter values within reasonable ranges. In this section, I examine analytical studies that model R&D activities in the pharmaceutical sector differently than Simpson et al. The question is, How do different assumptions regarding the R&D setting influence the value of genetic resources compared to the findings in the SSR model?

There are several studies that make use of an analytic framework in order to describe the value of biological resources in the search for new genetic information in drug development (Craft and Simpson 2001; Goeschl and Swanson 2002a; Kassar and Lasserre 2004; Polasky and Solow 1995; Rausser and Small 1998; Sedjo and Simpson 1996). Some of the studies support their theoretical results through the use of numerical simulations but none of the studies translate the results into monetary units as Simpson et al. (1996) and Rausser and Small (2000) do.

Most of the studies represent *static partial equilibrium* models. Only the studies by Goeschl and Swanson (2002a, 2003a) and by Kassar and Lasserre (2004) directly reflect *dynamic aspects*. The model in the studies by Goeschl and Swanson includes interactions between different stages of a vertically integrated industry. However, that model only represents the research process in a reduced form, while other studies provide a more detailed description of genetic resources as research options (e.g., Polasky and Solow 1995). Further studies place the emphasis upon a more sophisticated modeling of the *market structure* in the R&D sector, i.e., a description of the *R&D competition* (Craft and Simpson 2001; Rausser and Small 1998).

It is not helpful to give a detailed, formal introduction to each of the different modeling approaches. Instead, I identify two major areas that the selected studies address differently than Simpson et al. (1996). These are

- the economic properties of genetic resources in R&D, and
- the competition among firms conducting R&D (in the biotech sector).

In the following, I discuss how the different assumptions influence the value of a marginal genetic resource.

Substitution between Different Species

In the SSR model, the testing separates genetic resources into beneficial species and nonbeneficial species. Beneficial species are assumed to be perfect substitutes in that each of them yields a *benefit of equal size* for the researcher. Consequently, all beneficial species except for the first one to be discovered are redundant and yield no additional values. Sequential testing is terminated accordingly after the first beneficial species is found.

Alternatively, it can be assumed that the net benefit or value each beneficial species yields varies due to differences in quality or prospecting costs (Kassar and Lasserre 2004; Polasky and Solow 1995). Polasky and Solow (1995) develop a simple static model that describes the value of a collection of species, whereas a differentiated but uncertain value is assigned to each of the species. They demonstrate that the value of a given number of species that are imperfect substitutes is actually greater than the value when all species are perfect substitutes. Similar to the findings by Rausser and Small (2000), differences in the value derived from an individual species can lead to a larger value of the overall collection of species.[126]

In addition to substitution between different species, there is *substitution* between natural materials and *synthetic materials*. The SSR numerical simulation assumes that the proportion of new pharmaceutical products derived from natural products to products derived entirely from synthetic material is constant over time (1:3). In reality, this proportion changes over time if there is a shift in relative factor prices for natural products. Such a change can occur, for example, as a consequence of technological change, leading to relative quality improvement for either natural or synthetic material. In the end, this would also influence the value of a marginal species. However, the literature has not investigated this aspect.

Correlation with Respect to Success Probabilities

The SSR model relies upon the assumptions that (1) the probabilities of the R&D success of an individual *species* in different *research projects* are perfectly correlated. This means that they are identical across different classes of projects. At the same time, it is assumed that (2) the probabilities of success between species in an individual project are uncorrelated.

Rausser and Small (2000) briefly discuss whether the assumption of correlation across projects is reasonable. While examples of research on pharmaceuticals in unrelated therapeutic classes may provide intuitive counterarguments, the authors argue that path-dependent learning may support a positive correlation: for example, the detailed examination of an individual species in one project may provide insights that are useful in other projects. As mentioned earlier, Rausser and Small consider genetic resources that are differentiated with respect to the success probabilities. The SSR model more rigorously assumes that it is not possible to discriminate between any species in any projects, i.e., the R&D result in one project carries no information with respect to the success probabilities in all subsequent projects.

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[126] Rausser and Small (2000) consider *observable* ex ante differences in success probabilities, while Polasky and Solow (1995) consider *uncertain* differences in the value that are revealed ex post after the testing process. Uncertainty in turn is described by a known distribution function.

Polasky and Solow (1995) study the impact upon the value of a genetic resource/species when individual success probabilities within a research project are correlated. They assume that correlation depends upon *genetic similarity* between species, which is in turn measured by genetic distance. Genetically similar resources are supposed to share a similar probability of being a carrier of promising genetic information. The authors demonstrate that by introducing correlation into their model, the uncertainty with regard to the number of beneficial species in a given collection of species increases. In other words, the private value of the collection of species decreases compared to a situation with no correlation between species.

Correlation finally causes a given collection of species to yield a higher value the more genetically diverse the included species are. Furthermore, the value of a marginal species is not so much determined by the number of collected species, but rather by its genetic similarity to these collected species (Weitzman 1998).[127] Consequently, the value of specific genetic resources may not primarily be determined by the unit costs of testing and market revenues attainable in the sector, but rather by the genetic diversity/dissimilarity it adds to a given collection.

Option Values

In the SSR framework, R&D firms have complete information on what they can expect from a collection of genetic resources in all forthcoming research projects. There are uncertain but virtually no unknown future values of genetic resources. More specifically, researchers do not learn about how promising individual species are for future projects. Their private value is completely represented by the direct use value. Any option value component is not considered explicitly.

Arrow and Fisher (1974) and Henry (1974) first studied option values in the context of preserving biodiverse habitats. Fisher and Hanemann (1986) relate this analytical approach to the preservation of species habitat, whereas the role of species in pharmaceutical R&D is directly pointed out.[128]

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127 Polasky and Solow (1995) describe how correlation between species and imperfect substitution between beneficial species can be combined and what implications arise therefrom. Suppose that a specific beneficial species is shown to yield some value. Intuitively, further testing may focus upon close (and therefore genetically similar) relatives of that species. However, if the values of beneficial species are positively correlated and correlation increases with decreasing genetic distance, close relatives may only yield a similar value. Accordingly, testing may partly focus upon species that are less closely related.

128 While the model framework by Arrow and Fisher (1974) and Henry (1974) is considered virtually identical, the former use the term "quasi-option value," while the latter uses the term "option value" (Fisher and Hanemann 1986). It is worth mention-

Based upon the study by Kassar and Lasserre (2004), let us describe option values of species in the following way: an option value is assigned to a species that is presently not used because other species already or more effectively fulfill a desired biochemical service in R&D.[129] Since the *unused species* may sometimes fulfill the same service, preserving that species generates certain value as well. More precisely, in an environment of changing technologies and changing available endowments, the future may reveal that the previously unused species can fulfill the service in an efficient way. By not preserving it in the present, it may become extinct and its genetic information may be irreversibly lost. However, preserving unused species generally comes at a cost. Kassar and Lasserre (2004) investigate the demand for the species' preservation from the point of view of the R&D industry, which is facing uncertainty regarding future values of the genetic information.

Without addressing species' habitats explicitly, they study option values and the resulting implication for species preservation by applying a real option approach. Their model considers two beneficial species that both have the ability to fulfill the biochemical service in question. Consequently, a positive but differentiated use value is assigned to each of them. Given present technology, only the species with the comparatively higher use value is employed, the other remains unused initially. Due to technological changes, the use values of the individual species are subject to periodical changes. Therefore, it cannot be ruled out that it may be profitable to switch to the previously unused species at some point in the future.

Given the information available on the use values, the question is whether to allow the presently unused species to disappear or not. In Kassar and Lasserre's model, the answer depends primarily upon the current value of the unused species relative to the current value of the species in use. In addition, the authors formally show that increasing the (exogenous) volatility of periodical changes in the values of both species leads to an increase in the demand for preservation. At the same time, the correlation between the currently observed values of the used and unused species reduces the demand and makes abandonment more likely (Polasky and Solow 1995). Applying the formal results of the two-species model in a more general context, the value of a given collection of species is relatively larger when option values for presently unused species are taken into account.

Considering the SSR model, there are in fact no unused species: each project of sequential testing basically considers all existing species as available research options. In this respect, the private value is not understated by neglecting the

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ing that Dixit and Pindyck have introduced a different approach to option values in the context of business decisions on investments (Mensink and Requate 2005).

129 See also the discussion on genetic resources as imperfect substitutes (Kassar and Lasserre 2004; Polasky and Solow 1995).

option value component. In contrast, it is questionable as to whether R&D firms have access to all existing species when conducting R&D. This aspect is addressed in the discussion below on individual R&D firms and the aggregate industry. Finally, as long as no exclusive rights on potential future use are defined, option values from preserving species represent a public good (Weisbrod 1964). In this respect, option values would not add to the private value of genetic resources anyway.

While my survey so far has focused upon the economic properties of genetic resources in R&D, I now investigate in more detail how *market structure* and *R&D competition* are implicitly represented in the SSR and how alternative assumptions would influence the (numerical) value of a marginal species.

R&D Competition and Exclusive Uses

Rausser and Small (1998) argue that in the SSR model, competition between R&D firms on the demand side for genetic resources is neglected. In order to model competition, it may be assumed that there are several R&D firms, each possessing a collection of (partly) identical genetic resources and competing in a patent race to develop a new pharmaceutical.

Considering a specific untested genetic resource, an R&D firm fears that, if that resource indeed carries certain valuable information, a competitor may also be in possession of it and will discover and enhance the same genetic information. This reduces the chance that the R&D firm will be granted a patent right. Consequently, the ex ante value of an untested genetic resource to an individual R&D firm is larger if that firm possesses the *exclusive right* to conduct research with it. This applies independently of whether that resource ex post contains valuable information or not. Therefore, an R&D firm may generally be willing to pay a *premium* for the exclusive use rights to a specific genetic resource.

In the end, this demand for exclusiveness is actually driven by the property of nonrivalry in the use of genetic information. When there are many individuals of a biological species that carry the same genetic information, the testing of an individual in its material form does not have an impact upon the use of the information by testing any other individual.

Exclusiveness not only matters in the context of competition in a single targeted research project but also and more generally against the background of competition in the markets for final products, i.e., R&D outputs. Every untested genetic resource can potentially provide genetic information leading to the development of a new product that renders the products presently sold in the market obsolete. Accordingly, untested genetic resources as research options threaten the position of those firms that have managed to attain a certain market share of the output market. If exclusive use rights to genetic resources are granted, each

R&D firm then has an incentive to purchase them for *defensive reasons* (Rausser and Small 1998).[130]

Based upon these considerations, R&D firms apparently display a relatively higher willingness to pay for an individual genetic resource than implied by the SSR model, which does not consider this strategic aspect. According to Rausser and Small (1998), each R&D firm will offer money until the redundancy cost of a marginal species in its portfolio equals the benefit from keeping that species away from the portfolios of its competitors. However, this conclusion relies upon the assumption that exclusion by the providers of genetic resources is manageable. While the granting and monitoring of exclusive access can be challenging in practice for endemic as well as nonendemic species, the merit of the study by Rausser and Small is that it points out that providers have more options than just to destroy resources or grant access to everyone (Rausser and Small 1998). As long as enforcing exclusive access increases the willingness to pay on the demand side and as long as there are no excessive transaction costs of exclusion, the providers prefer this regime, since it more than likely offers them a relatively higher payoff.[131]

R&D Competition and Final Products as Imperfect Substitutes

The SSR model, as well as Rausser und Small's model (1998) relies upon the simplifying assumption that the revenues, R, in the case of a success in R&D are constant and, thus, independent of the number of products on the market or the number of new developments.[132] In other words, a product will not reduce the demand for other products and leave prices and sales unchanged; both studies consider R&D outputs as entirely unrelated (Craft and Simpson 2001).

By contrast, Craft and Simpson (2001) employ an alternative modeling approach in which products on the output market are imperfect substitutes. They adopt two models from the literature, the Salop model and the Dixit–Stiglitz model, that describe consumer behavior when differentiated goods are supplied on the market. It turns out that, for both models, revenues for a pharmaceutical

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130 Rausser and Small (1998) derive their findings from a specific model of R&D competition: a finite number of R&D firms competes for a patent by testing research options. Exclusive rights to these options are a priori auctioned off by the suppliers of genetic resources. For simplicity, firms cannot draw upon their own ex situ endowments, i.e., there is no overlap between the individual collections of the research options.

131 It is required that that the revenues a supplier can earn from selling an *exclusive* access right to a *single* R&D firm are greater than the those from selling access rights to many firms

132 The SSR model, furthermore, assumes that the number of research projects conducted is constant, i.e., that λ is exogenously given.

on the market decrease when new products are introduced. The connection to the input market for genetic resources is established in the following way: when the number of (expected) new products increases with the number of species tested and this influences the revenues attainable on the market, the private value of species decreases. Based upon this, the value of a marginal species derived from the SSR model somehow tends to be overstated.

Individual R&D Firms and the Aggregate Industry

Throughout their paper, Simpson et al. are not explicit regarding the market structure of the R&D industry; they especially do not address the number of firms as buyers of genetic resources and how this figure influences the revenues a provider can earn. It seems that the authors implicitly assume that either (1) the entire industry *acts collectively* to preserve genetic diversity as a *club good* or that (2) the *individual demand* for bioprospecting area is *perfectly elastic* and, therefore, the individual maximum willingness to pay is identical for all R&D firms and is equal to the willingness of industry as a whole. The first storyline implies that though individual firms compete in the market for final products, there is strong incentive for each individual firm to participate in collective action. However, the authors do not investigate under what conditions this prerequisite can indeed be satisfied. With the second storyline, the number of firms in the market is irrelevant for the value or market price of a marginal hectare of biodiversity land.

The following own considerations support the hypothesis of a perfectly elastic demand in the model: suppose that individual firms purchase species for research purposes. In the SSR model, the value of a marginal species is conditional upon the total number of available species that represent research options for the individual firm. This value is constant over the whole range of species that a firm can access. Suppose for simplicity that the firms possess identical technologies. Since the value of a marginal species in the SSR model is conditional upon the *total* number of species and the *total* number of expected new products,[133] it is implied that each firm is able to access the entire collection of species, N. If this is the case, the value each firm assigns to a marginal species is identical for all firms.

In equation (3.28),[134] the firm's willingness to pay for a hectare of biodiversity land is equivalent to the species productivity of that land, i.e., elasticity times density, $z \times D_i$, multiplied by the "price" of a species, i.e., the derived value of a marginal species. The species productivity is site-specific and, thus,

133 This is indicated by the assignment of parameter values to the exponent, n, and the calibration of the parameter λ.

134 In Simpson et al., this is equation (10) in connection with equations (11) and (12).

also identical for all R&D firms. Consequently, all firms also have the same inelastic willingness to pay for the right to conduct bioprospecting in a given hotspot; the market demand is represented accordingly.

The definition of constant and identical values for each firm carries the strong assumption that all firms can access virtually all species. Thus, the derived value refers to the opportunity cost of irreversibly losing the first species. Let us alternatively suppose that each of several R&D firms on the market only has access to a limited share of the existing species richness: each firm has already stored certain species in its *own ex situ collection*, which is used exclusively in the firm's research projects. Accordingly, each firm conducts bioprospecting purely for the purpose of *expanding the in-house collection* (see Section 3.3.2.2). As a consequence, the individual willingness to pay would no longer be constant, but would depend upon the number of species stored in their own ex situ conditions. When the firms differ with respect to their ex situ resources, their willingness to pay also differs because, in this setting, the number of resources already stored influences the value a marginal species displays from the perspective of an individual firm. The individual demand for species would no longer be inelastic.

Furthermore, since, on the one hand, the probability of a success in R&D increases for each firm with the number of species it possesses and since, on the other hand, the total number of new products is assumed to be constant, increasing efforts in the on-site collection by one firm would reduce its competitors' willingness to pay for bioprospecting opportunities. This is because if one firm expands its collection for use in R&D, the chances for the other firms to yield a success in R&D decrease, ceteris paribus. The value of marginal species from the perspective of industry would be influenced by the aggregate size of the individual collections.

When the model framework is modified in the way discussed, (1) the number of firms on the market and (2) the size of their individual private collection of genetic resources need to be defined explicitly. If in this regard the market demand for species displays some elasticity, i.e., the demand curve is downward sloping, the impact upon the profitability of the market supply of in situ genetic resources that results can, however, not be generalized. This is because when the R&D industry shows a somehow elastic willingness to pay that remains at a relatively low level, the market revenues for the supply side need not increase in comparison to a situation with a perfectly elastic market demand. When regarding an individual in situ provider, the revenues he can earn would depend upon the absolute level of that willingness to pay, the elasticity with which it declines when the R&D firms increase their ex situ collection, and the number of firms that contract him. These considerations are not addressed by Simpson et al. and offer an opportunity for future analytical work.

Table 10:
Changes in the Value of a Hectare of Bioprospecting Habitat: Summary of Results from a Qualitative Analysis of Alternative R&D Settings

Characteristic	Study	Impact on land values[a]
Substitution: natural vs. synthetic material		( )
Heterogeneous genetic resources (w.r.t. ex ante success probabilities)	Rausser and Small (2000)	(↑)
Correlation between species (w.r.t. success probabilities)	Polasky and Solow (1995) Kassar and Lasserre (2004)	↓
Option values	Fischer and Hanemann (1986) Kassar and Lasserre (2004)	(↑)
R&D competition: exclusive access and use rights	Rausser and Small (1998)	↑
R&D competition: R&D outputs as imperfect substitutes	Craft and Simpson (2001)	↓
Individual demands by firms and aggregate demand	Own reflections, Simpson and Sedjo (1996)	( )

[a]Changes in land values relative to findings in the SSR simulation.

Finally, in a study subsequent to Simpson et al. (1996), Simpson and Sedjo (1996) analyze the relationship between individual payoffs from R&D and aggregate payoffs on an industry level in detail. However, they change the formal framework by assuming the *simultaneous testing* of species and they do not show whether their findings can be applied to the model of sequential testing.[135]

Table 10 summarizes the findings of the previous discussion. The right-hand column indicates how different assumptions regarding the R&D setting as presented in the SSR model influence the derived value of a hectare of land. It is shown that by introducing alternative characteristics of the R&D process, the private value may change in different directions. Thus, it cannot be concluded whether the results by Simpson et al. (1996) actually overstate or understate the actual but unobservable value.

135 The value of the collection of m species turns out to be $V(m) = R[1-(1-p)^m] - mc$. The notation follows that used in Simpson et al. (1996).

3.4.3 Alternative Ecological Environments and Their Impact upon the Private Resource Value

In addition to modifications of the economic representation of the R&D process and R&D competition, variations in ecological parameters and processes in the in situ environment can be introduced. In Section 3.4.3.1, the analysis still remains within Simpson et al.'s model framework. I present some reflections on the impact of alternative assumptions upon promising species, species richness, and density within the hotspots. Section 3.4.3.2 discusses the role of adaptive processes in an evolutionary biological system for the private value of genetic resources in more detail. In this context, the investigation goes beyond the SSR framework and considers an alternative modeling approach on biological adaptations to newly developed biotechnological products as represented by Goeschl and Swanson (2002a, 2003a, 2003b).

3.4.3.1 Alternative Ecological Characteristics of In Situ Habitats

First, I briefly investigate whether it is adequate to define species as research options as is done in the SSR model and discuss the role of endemic versus non-endemic species. Subsequently, I develop my own simple approach to capture habitat heterogeneity within hotspots.

Species as Basic Units of Research

In the SSR model, as well as in other studies, species are considered "basic units of genetic differentiation" (Simpson et al. 1996: 168). In contrast, empirical evidence on pharmaceutical R&D suggests that the research focus is on *biochemical compounds* as the units of research (ten Kate and Laird 1999: 40ff.; WBGU 2000: 69ff.).

Compounds are derived from extracts of biological material; more than 100 compounds with biological activity may be contained in a plant extract (ten Kate and Laird 1999: 51). From this perspective, the total number of higher plant species may understate the actual amount of available research options. However, not all plant species may yield as many compounds that are of interest to pharmaceutical R&D.[136]

[136] Some studies use empirical estimates for success probabilities and apply these figures to the set of available research options (Barbier and Aylward 1996; Artuso 1996a; Pearce and Puroshothaman 1992). In this regard, it is important how the set of options is defined and what is considered as the basis of the success probability. As argued in PhRMA (2003), different probabilities can be assigned to material in the different stages of the screening process. In the SSR model, this problem is circumvented in that the authors argue that an upper limit of the value of a marginal species

According to the WBGU (2000: 40), approximately 25,000 *medicinal plant species* are currently in use, 5,500 of which have been almost completely investigated. Future bioprospecting efforts may therefore focus upon approximately 20,000 previously unused medicinal plants. Depending upon the number of promising compounds contained in these species, the set of research options would increase or decrease relative to the number of options assumed in the SSR numerical simulation.

Endemism and Supply Side Competition

In the numerical simulation by Simpson et al., only habitats for *endemic species*, i.e., plants that only grow within an identified hotspot and nowhere else on earth, yield a positive value for the landowners.[137] The number of *nonendemics* within a hotspot does not influence the willingness of the R&D industry to pay. In contrast, the total number of species (250,000) enters the numerical simulation, i.e., it is assumed that nonendemic species can be used well in the R&D process. This can be interpreted such that it is implicitly supposed that R&D firms purchase nonendemics (nearly) free of charge. The reason for this is implicit competition between suppliers of nonendemic in situ genetic resources. Competition drives the price for genetic resources to a marginal cost that is seemingly close to zero for a single nonendemic genetic resource (Aylward 1993; Artuso 1997). In practice, this assumption is quite rigorous. Moreover, suppliers may indeed obtain a somewhat positive market price for nonendemics.

Heterogeneous Units of Land

Equation (3.28) indicates that the elasticity, z, and the species density, D_i, are the ecological characteristics of the bioprospecting land that control the firms' willingness to pay. Since z is assumed to be identical across the hotspots, differences in land values between hotspots only result from differences in the observed *density of endemic plant species* per hectare, D_i. Because Simpson et al. derive a unique willingness to pay for each individual hotspot, this density is implicitly assumed to be identical across the entire area of each hotspot. In other words, each hotspot is a homogenous habitat, not in the sense that each species is evenly dispersed across the area but that the total number of species in each arbitrary parcel of land is identical.

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is described; a hypothetical success probability that maximizes the value is used in the simulation.

137 The same may apply to endemic species outside the hotspots. However, this aspect is not discussed by Simpson et al. any further. Since species density per hectare in regions outside the hotspots is assumed to be lower, the willingness to pay for access to these areas is also below the values derived for the hotspots.

This implicit assumption may be relaxed, since a hotspot covers a quite large area; more precisely, it is on average 8.5 million hectares in Myers (2000) and 4.1 million hectares in the original classification.[138] Similar to the arguments in Rausser and Small (2000), researchers may observe that subregions within a hotspot display different densities in species richness. Bioprospecting activities are then likely to focus upon the most promising sites, i.e., "*the hotspots within the hotspots.*" Since this aspect has not yet been considered in the literature, I provide my own numerical considerations by adopting the SSR model framework.

Data on the spatial structure of diversity within the hotspots is neither available in the study by Myers, cited in Simpson et al., nor in the study by Myers et al. (2000). Yet, to give an impression of how heterogeneity can influence the results of the numerical simulation, let us suppose that a *subregion* is located within the hotspot. Given the data for a hotspot, i, regarding its total area and the number of endemic plant species hosted therein, let us describe the subregion by e, which denotes the percentage of endemic plant species in the entire hotspot that can be found in the subregion, and s, which is the size of the subregion as a percentage of the entire hotspot. The subregion's species density in endemic plants is $\hat{D}_i = D_i(e/s)$.

To describe the willingness to pay for a hectare of bioprospecting land in a subregion richer in biodiversity, it is of importance that the value per hectare, derived in equation (3.28), increases linearly with the species density, D_i. For this reason, the land value derived for the entire hotspot, i, only needs to be multiplied by the ratio e/s in order to obtain the value for a hectare of land in the subregion. All other variables, such as the derived value of a marginal species, remain unchanged.

Suppose, for example, that it is possible to demarcate a subregion of 20 percent of the total area that hosts 40 percent of all endemic plants in the hotspot. Accordingly, the willingness to pay for a hectare of preserved land located in that subregion is twice as large as the value derived for the total hotspot area. In the same respect, subregions may be identified that host few endemic plants and where the willingness to pay is less than the average for the total hotspot. Nevertheless, it can be concluded that if identified land values generally fall below the level needed to compensate for alternative land use, as implied by Simpson et al., more narrowly defined hotspots are more likely to represent sites in which incentives for conservation may work effectively.

To analyze this in more detail, I consider only the hotspot with the highest species density per hectare, i.e., the *Western Ecuador hotspot* (250,000 hectares

[138] Note that the extent and boundaries of hotspots are not determined on economic grounds. They are based upon biological information only.

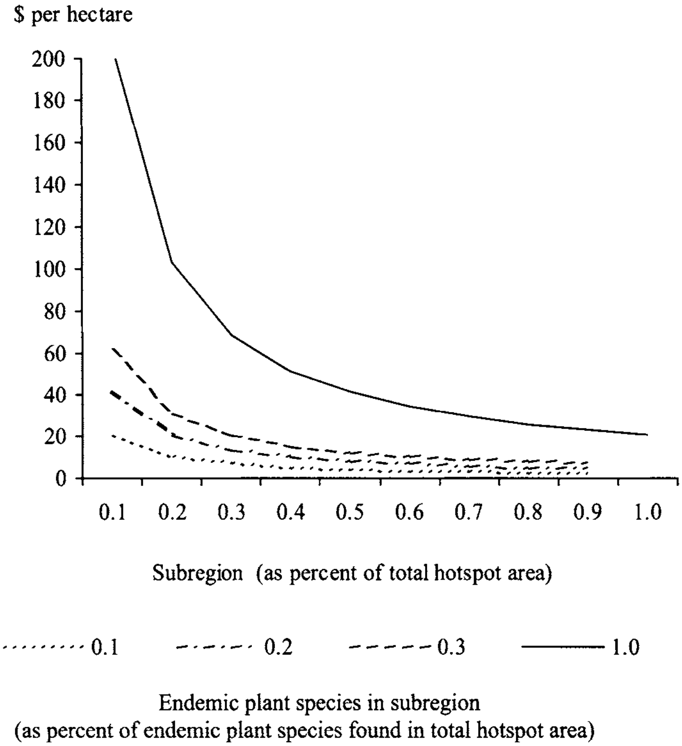
with 2,188 endemic plants). Remember that Simpson et al. derive a land value of \$20.63 per hectare. Since this value refers to the entire hotspot area, I am interested in the values that can be assigned to subregions of a varying size and level of endemism. For this purpose, I fix the parameter e, which determines the number of endemic species in the subregion, at a certain arbitrary level. I describe how land values change when the size of the subregion, which is described as share s of the total hotspot area, changes.

The numerical results are presented in Figure 13. By definition, an extreme subregion representing the entire hotspot, i.e., $e=1$ and $s=1$, yields a value of \$20.63 per hectare. If researchers can find the same endowment of endemic species within a smaller region, i.e., if s decreases, the land value for a hectare in that region increases. However, it may be very optimistic to expect even a very diverse subregion to host all of the endemic species that occur in its neighboring regions. Let us, therefore, assume comparatively low values for e, i.e., $e=\{30\%,20\%,10\%\}$. It turns out that under these conditions, a land value of more than \$20 per hectare is only obtained for subregions covering less than 30 percent of the total Ecuador hotspot.

In this exercise, I have varied e and s in order to derive land values for preserved bioprospecting land. To access the profitability of a supply of bioprospecting opportunities, let us proceed the other way round, i.e., set the value of preserved bioprospecting land equal to the value of land in the alternative use and fix the level of e in order to derive target values for s, the relative size of the subregion. The resulting combinations of parameter values can be accessed with respect to their consistency.

To calculate the relative size of the hypothetical subregion, I extend the term in equation (3.28) that describes the value of marginal land by the factor e/s and set the value equal to \$400 per hectare, which I assume to be the value of land in the alternative use. This figure is taken from Chomitz et al. (2005), who derived it for forests in the Brazilian region of Bahia. For simplicity, let us define that the R&D firms' maximum willingness to pay for a hectare of land must be equal to \$400 in order to render preservation profitable. Furthermore, I again consider the Western Ecuador hotspot and assume that an inner hotspot hosts $e=\{5\%,10\%,15\%\}$ of the entire hotspot's endemic species. Solving s and applying the numerical solution to the combination e and s, which yield the target value, the corresponding subregion, for which conservation is profitable, covers an area (in hectares) of $s\times 250{,}000=\{1934,1289,645\}$. These areas would, in turn, correspond to a hypothetical species density, i.e., richness per hectare, of 0.1697. Compared to the values of species density in the original hotspot study (0.0088 at maximum), the hypothetical value that needs to be attained in order to enable a profitable supply of bioprospecting areas, is very large and would at best apply to selected, small natural areas.

Figure 13:
Land Values of Subregions within the Western Ecuador Hotspot



To conclude, heterogeneity within the hotspots may indeed lead to land values that are higher compared to the values estimated in the original simulations. This implies that neglecting habitat heterogeneity for large hotspots causes predicted land values for certain biodiverse subregions to be somewhat understated. However, through the use of numerical simulations, suggestive evidence is found that implies that by taking into account habitat heterogeneity, land values do not necessarily increase to a sufficient extent to render preservation for the market supply of genetic resources profitable.

Finally, the modification presented does not include endogeneity with respect to the total species richness, $n = N$, that influences the value of a marginal species. More specifically, N will surely be decimated if only the subregion is preserved and the rest of the hotspot is degraded. When N is about to decrease, i.e., species richness declines, the value of a marginal species and, hence, the willingness to pay for a hectare in the subregion increases ceteris paribus.

3.4.3.2 Repercussions from the Ecological System

So far, the model underlying the calculations is essentially static. Goeschl and Swanson (2002a, 2003a and 2003b) argue that particularly *adjustments within the ecological system* have a significant impact upon production that relies upon biotechnology. To their mind, this impact should be taken into account when attempting to evaluate the genetic resources for R&D and the conservation of their habitats. This point of view is in line with the reasoning on the economic-ecological integrated modeling described in Section 2.3.1. The focus falls upon the *biological responses* to human resource management.

In order to investigate this interaction, I first stylize the major aspects of the repercussions from the biological world upon the effectiveness of biotechnological products that rely on genetic resources and describe how Goeschl and Swanson conceptualize these repercussions in a modeling framework. I then assess the importance of repercussions upon pharmaceuticals on the basis of anecdotal evidence. Finally, I discuss possible extensions of the SSR model in order to incorporate biological repercussions.

Antibiotics, HYVs, and Pesticides: Evidence on Repercussions

The impact of biological responses becomes most visible in the fact that, compared to innovations in other industrial sectors, the output of biotechnological R&D often does not represent durable solutions to problems of human life but rather loses its effectiveness over time. Examples are applications of *antibiotics* in the health sector, the use of *high yielding varieties* (HYV) that contain pest resistance, and the use of *pesticides* in agricultural production. The reason for this loss of effectiveness is that antibiotics, as well as modified crop varieties or pesticides, represent human interventions in the ecological system that automatically involve natural responses that diminish and erode the desired effectiveness of the intervention (Goeschl and Swanson 2002b, 2003b; Munro 1997; Swanson 1996).

Such adjustments can be explained on the grounds of biological science: let newly developed biotechnological products be represented by resources that initially demonstrate a relatively high effectiveness against biological predators. Since these resources contribute more to human well-being than other resources do, man favors them over the others, with the consequence that they are widely spread across the economic-ecological system. However, this selection simultaneously favors those individuals among the predatory organisms that are not susceptible to the effects of the beneficial resources or that manage to break the defenses of these resources. In this regard, biological predators "prosper by reason of our own choices" (Goeschl and Swanson 2003a). The resulting *erosion of the effectiveness* of biotechnological products seems inevitable. Furthermore, the speed and the scale of these adjustments is endogenously driven by the extent of human interference in the biological system (Goeschl and Swanson 2003b): an accelerated pace of new biotechnological developments that succeed each other and their widespread use is assumed to cause parallel responses in the form of increasing biological adaptations.

Modeling Repercussions in an Economic-Ecological Framework

To repeat, the widespread use of a biotechnological product is crucial to biological adaptations. To model repercussions in an analytical framework, Goeschl and Swanson use the term "widespread" in its spatial meaning: in their basic model, the extensive use of a prevailing biotechnological product is represented by the *allocation of land* among different uses. More precisely, the authors assume that biotechnological innovations dominate the use of land that is directly productive from the human perspective, and thus define a monoculture regime. Other land that has been set-aside constitutes the biological reserves. Given the positive relationship between the extensive use of biotechnological products and the *degree of biological adaptations*, the modeling implies that biological reserves do not merely serve as the in situ gene pool for biotechnological R&D, but also reduce the emergence of adaptations to man-made innovations by reducing the land in productive but monoculture uses (Goeschl and Swanson 2002a; Laxminarayan and Simpson 2002).

To conceptualize the interactions within the biological system, as well as the between biological and economic system, Goeschl and Swanson develop the approach of "adaptive destruction" to biotechnological innovations.[139] For this purpose, the vertically integrated, multifirm biotechnological industry is described by a three-tiered structure. The analytic framework represents an adaptation of Aghion and Howitt's *model of endogenous growth* (1992).[140] While their original model only considers "creative destruction," i.e., man-made innovations that make existing products on the market obsolete, Goeschl and Swanson (2002a, 2003a) additionally introduce the possibility of "biological innovations" that reduce productivity in the production of the final good.[141] The occurrence of these

[139] The authors do not explicitly model biological interactions as intertemporal adjustments. However, the derived equilibrium terms serve to explain the expected changes in the trajectory of future production possibilities that result from the biological adaptations.

[140] The original model distinguishes unskilled, skilled, and specialized labor as the only production inputs (Aghion and Howitt 1992). Goeschl and Swanson (2002a, 2003a) translate this classification into a classification of land endowments—with land for the production of intermediates, land for the final good production, and undisturbed natural land (biological reserve).

[141] A key variable in the model is the productivity factor in the final production. In the original model, this variable increases with every innovation that is made, i.e., a

productivity-reducing biological adaptations depends upon the land allocation, i.e., the size of the biological reserve that is withheld relative to the size of the land cultivated in an ostensibly productive but biologically homogenized way.[142]

Against this background, Goeschl and Swanson (2002a) criticize the modeling approach based upon the concept of redundancy proposed by Simpson et al. (1996) and Rausser and Small (2000). The approach is criticized mainly not because it disregards the positive externality from undisturbed biodiverse areas on the (long-term) effectiveness of R&D products but, more importantly, because it assumes a finite and exogenously given number of biological problems that are addressed by biotechnological R&D and (lastingly) solved.

Health Hazards due to Repercussions from the Biological World

In the following, I discuss whether the criticism of the SSR model with respect to its negligence of repercussions from resource management upon the value of genetic resources is justified.

In this context, it first has to be noted that the SSR model explicitly describes R&D in the pharmaceutical sector only. Accordingly, it can be asked how important the biological repercussions stylized by Goeschl and Swanson upon the health sector are. Empirical evidence shows that primarily *antibiotic drugs* are subject to biological adaptations described by the spread of antibiotic resistance (Smith et al. 2005). Furthermore, it is indicated that in contrast to antibiotics, many other pharmaceuticals have retained their effectiveness over many decades, even centuries. Well-known examples are *acetylsalicylic acid* or products of *botanical medicine*. The reason for the sustainable effectiveness of certain pharmaceuticals is that they address health problems that are not due to external biological predators but that have their causes in private circumstances, such as individual choices, lifestyle or genetic predisposition. Examples are health problems due to ageing, malnutrition, mental disorders, or hereditary diseases (Chen et al. 1999).

Based upon this, I examine the importance of antibiotics in the market for pharmaceuticals in order to assess the role of biological adaptation for R&D and the modeling of R&D. Regarding the empirical data on current global sales, it turns out that the pharmaceuticals other than antibiotics are the major sources of

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"quality ladder" that can be climbed is assumed (Aghion and Howitt 1992). In contrast, Goeschl and Swanson (2002a, 2003a) allow for the possibility that the gain in productivity is offset because of the occurrence of biological responses to biotechnological products.

142 To discuss the theoretical results of the model in more detail, Goeschl and Swanson (2002a, 2003a) perform numerical simulations. The figures they derive are, however, not presented in monetary units and are therefore not comparable to the numerical results discussed in the previous sections.

revenues in the sector (IMS 2005). In other words, when comparing antibiotics and other pharmaceuticals, the former are apparently of a lesser importance to (profit-oriented) R&D.

With regard to the criticism of the concept of redundancy, I therefore conclude that it has to be considered that the concept refers to pharmaceutical R&D, and that, in my considerations, it cannot be confirmed that it would be inappropriate to neglect the role of biological adaptations, since they are only of minor importance in this sector.[143]

Allowing for Biological Repercussions in the SSR Model: A Discussion

In spite of this assessment, the pharmaceutical industry indeed develops new antibiotics. Since the problem of biological adaptations (antibiotic resistance) actually exists for these products, I provide an outlook on how changes in the modeling framework by Simpson et al. may have an impact upon the derived private value of a marginal species.

The SSR model relies upon the assumption of a constant number of biological problems, which implies that predatory organisms do not develop resistance to newly developed pharmaceuticals, i.e., antibiotics in the further context, within the period of patent protection. Thus, the effectiveness of those pharmaceuticals is maintained and, therefore, the expected revenues, R, are independent of any repercussions. Emerging adaptations also do not change the size of the annual research portfolio represented by λ.

To allow for a more noticeable impact of biological adaptations in the model, I relax the simplifying assumptions one by one. For discussion, suppose that there are increasing biological adaptations. The following drivers represent the potential economic adjustments resulting therefrom:

- Increasing biological adaptations lead to a decreasing period of time in which a newly developed pharmaceutical demonstrates its positive effects in medication. To close the resulting gaps in medicinal treatments that happen earlier and more frequently more R&D is needed. This means that R&D firms expand the *size of their research portfolios*, which effectively increases the value of a marginal resource.
- Decreasing the effectiveness of pharmaceuticals can lead to parallel responses on the demand side. If, due to increased biological adaptations, pharmaceuticals become obsolete more quickly, the expected revenues that a firm

143 Finally, the problem of resistance is in turn related to overuse in medication (Smith et al. 2005; Goeschl and Swanson 2002b). In contrast to the framework in Goeschl and Swanson (2002a), biological adaptations to these pharmaceuticals are not dependent upon the allocation of land and biodiversity conservation.

earns from bringing a new product onto market decrease. Consequently, the resource value decreases.

- Increasing adaptations that induce increasing research efforts may influence the *R&D costs*. For example, if the effectiveness of more and more antibiotic substances is exhausted, the search costs of finding new ones may rise. Rising research costs, in turn, reduce the net revenues for the R&D firms and therefore lowers their willingness to pay for genetic resources. Accordingly, the resource value decreases.[144] However, decreasing net revenues can also lead to changes in the market structure that, in turn, have an impact upon the revenues an individual R&D firm can earn on the final good market and, thus, lead to repercussions upon the firm's willingness to pay, as well as upon its bargaining power on the input market for genetic resources.

To resume, it is difficult to assess in what direction the impact of biological adaptations in the health sector would influence the value of a marginal species relative to the value derived in the SSR model.

Finally, Goeschl and Swanson (2002a, 2003a, and 2003b), who incorporate the biological repercussions in their approach, do not analyze questions regarding the private value of genetic resources. The authors focus upon the normative question as to whether the existing regime of property rights on genetic information guarantees an efficient allocation (see Section 3.1.2).

3.4.4 The Private Value of Genetic Resources in Agriculture

Aside from the pharmaceutical industry, the plant breeding industry is a major commercial user of genetic resources. I briefly describe the theoretical approaches to (private) values of genetic resources in agriculture and recapitulate the role of markets in the allocation of these resources.

Regarding the commercial use of crop genetic resources in the agricultural sector, there are numerous studies that investigate the economic value of these resources both upon a theoretical and empirical basis. These studies typically investigate the impact that the introduction of new crop varieties containing specific characteristics (traits) has on the average agricultural yield, the yield variances, and/or farmers' income (e.g., Smale et al. 1998; Evenson 1996).[145]

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144 Alternatively, an increasing scarcity of antibiotics may lead to rising market revenues for these products, so that the net revenues need not decline, i.e., the market environment on the demand side of genetic resources does not necessarily have to change because of biological adaptations.

145 Other values stem from increases in yield and income due to a diversification in the use of genetically diverse crops (e.g., Di Falco and Perrings 2003).

Consequently, the value of crop genetic resources that is described in these studies refers to the share of the social value of genetic resources that is appropriated by *farmers* as the commercial users of (modern) cultivars.

From a methodological perspective, most studies are based upon a *production function* approach. Other studies that use a *partial equilibrium framework* also take into account the benefits for the R&D firms and consumers and investigate changes in welfare on a national and international level (e.g., Frisvold et al. 2003; Falck-Zapeda et al. 2000): the study by Falck-Zapeda et al. (2000) analyses the worldwide surplus from introducing a specific transgenic crop variety and how this is distributed among consumers, farmers, the seed company, and the R&D firm as the inventor.[146]

Since the breeding materials predominately originate from ex situ facilities (see Section 3.2.2.3), several studies develop an analytic framework in order to determine the specific value of ex situ genetic resources in the form of gene bank accessions (Zohrabian et al. 2003; Koo and Wright 2002; Gollin et al. 2000). These studies typically describe the optimization problems of *gene bank managers*.

Both types of studies provide little information on the potential values that providers in in situ environments can appropriate. This is apparently because the studies implicitly assume that plant breeders obtain the needed materials from ex situ collections *free of charge*. Consequently, they do not consider how much an in situ provider can earn by supplying crop genetic materials.

Finally, the studies do not describe in detail how many genetic resources are used in a specific breeding process. Although genetic resources serve as carriers of these traits, the values derived in the studies are not usually assigned to single species or varieties, but rather are attributed to the available gene pool in general.

In Section 3.3.2.3, it has already been described why the current regime of *in situ management* of crop genetic resources does not seem to generate substantial incentives for the in situ preservation of biodiversity. In the context of on-farm conservation in a private sector environment, there are several studies that model farmers' decisions on crop choices on a theoretical basis (Heal et al. 2003; Weitzman 2000) or empirically investigate the actual decisions of the farmers (e.g., Brush et al. 1992). The question of revenues that farmers can obtain by exchanging seed materials with other farmers or by supplying them to users in the plant breeding industry is addressed.

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[146] It turns out that the R&D firm appropriates 21 percent of the surplus (or about $50 million) in the first year the crop is introduced (Falck-Zapeda et al. 2000).

3.5 Summary of Results

Regarding the search for the societal acceptable and sustainable management of biodiversity as a resource of global importance, this chapter concentrated on the role of markets for ecosystem services as a means to biodiversity conservation. As described in Section 2.3.3, the market mechanism can assist conservation efforts for the various ecosystem services of a different economic nature. The analysis in this chapter focused upon genetic resources as private natural resources that contain some private value beyond the resource-use value. The market allocation of these goods is subject to debates in academia and international policymaking. I considered four areas of research in genetic resources.

Properties of Genetic Resources as Economic Goods

By describing the commercial use of genetic resources, I showed that biological materials as carriers of valuable genetic information display specific economic properties that distinguish these resources from other renewable natural resources: these are (1) *uncertainty* with respect to the appropriable value and (2) *nonrivalry* in the use of the information contained. Furthermore, genetic information is replicated or synthetically reconstructed in in-house laboratories without the need for genetic material from the original in situ sources.

I presented empirical evidence that suggests that, in practice, these properties may not apply to every case: as illustrated by ex ante information (including traditional knowledge) regarding genetic material, uncertainty may be reduced. Furthermore, biochemical replication may be impossible for technological reasons. In this case, the commercial use of specific genetic information is bound to genetic material from the original in situ sources. These variations imply that the nature of genetic resources cannot be generalized, but rather that it is quite diverse and depends upon the prevailing conditions at a specific site and a specific point of time.

Given the economic properties and empirical description of commercial use in different economic sectors, I distinguished three *major types of use*:

- conventional natural-resource-like use,
- informational use in the pharmaceutical industry and other biotechnological applications, and
- informational use in the plant breeding sector.

I described each type of use in qualitative and quantitative terms.

The Allocation of Genetic Resources and Genetic Information and the Role of Property Rights: A Summary

The analysis continued with a description of the international allocation of genetic resources and the information embodied. I have shown how the allocation is governed by the special *allocation of access and use rights*. More precisely, the international community has implemented different property right regimes, both with respect to wild and improved genetic material.

As a result, these regimes cause genetic resources to be allocated in different but parallel-existing institutional settings. They are provided as impure public goods and/or private goods. Providers are profit-oriented landowners, ex situ suppliers, or public sector institutions. The specification of access and use rights is partly due to the economic properties (particularly with regard to nonrival use) and partly due to social constructs that are determined by policies and collective arrangements. The latter involves legal provisions that regulate the access to the resources and exclusiveness of their use. Figure 14 classifies the findings of the chapter with regard to the allocation of biological resources and information goods. Use externalities (rivalry) and exclusiveness serve as characteristics to distinguish the different segments of the allocation.

In addition to the classification by economic properties, I recapitulated the complex *international property rights regime* with respect to biological resources and its embodied genetic information. The findings are depicted in Figure 15. The starting point was the *CBD*, which is the dominant regime in biodiversity policy. This agreement provides countries with the sovereign right to control access to the genetic resources within their own borders and regulate use by means of national access laws and bilateral contracts. Since exclusive and enforceable rights to control the access and use genetic resources are a prerequisite for profit-oriented biodiversity conservation, the provisions of the CBD support a market-based approach to conservation. Furthermore, it turned out that the CBD primarily addresses property rights to genetic material but not genetic information, i.e., the CBD does not establish markets for genetic information.

By analyzing the international property right regime for genetic resources relevant to plant breeding (*PGRFA*), I illustrated that there are deviations from the principles of the CBD on major points. This is namely in the public domain status of some ex situ genetic materials, as well as a mandatory multilateral funding mechanism to preserve the world's gene pool as a global common. In addition, while the recent IT-PGRFA includes intellectual property right clauses, the use of genetic information is regulated outside this regime.

Figure 14:
Economic Properties of and Property Rights to Genetic Resources and Genetic Information: A Summary

		Genetic resources	Genetic information
Rivalry		Rival in its use	Nonrival in its use
Exclusiveness	Private property	Biological material	Isolated (and purified) genetic information (IP[a] protected). Synthesized genetic information (IP[a] protected).
	Common property[b]	Biological material (in Multilateral System)	Traditional knowledge. (Locally) applied genetic information.
	Open access[c]	Biological material (in open access regime)	Information for which IP[a] protection has expired. Knowledge in the public domain.

[a]IP stands for "intellectual property." — [b]Use rights are controlled by explicit rules and linked to membership/group affiliation. — [c](De facto) unlimited use/extraction rights, independent of membership.

Source: Wolfrum et al. (2001: 41), Stevenson (1991: 58), own representation.

Figure 15:
International Property Rights Regimes on Genetic Resources and Genetic Information: A Summary

	Genetic resources	Genetic information
Commercial use in general	*CBD* (1992) Sovereignty Bilateral benefit sharing	*TRIPS* (1994) Patents Other IPR
Commercial use in the agricultural breeding sector	*IT-PGRFA* (1996) Sovereignty plus Multilateral system ("Limited" common property) Multilateral a/o bilateral benefit sharing	*TRIPS* (1994) / *UPOV* (latest rev.1991) Patents Plant breeder rights Other sui generis system

Source: Virchow (1999b), own representation.

For both types of genetic resources, the use of the information embodied is determined in specific *systems of intellectual property rights*. I described how the *TRIPS* is the major regime in this context. It generally applies to the use of genetic information in all sectors. Only for the plant breeding sector does the *UPOV* regime allow for a sui generis IPR system in addition to the one arranged in the TRIPS.

The Question of an Efficient Supply of Genetic Diversity

Given the economic properties of genetic resource and genetic information, the question was posed as to whether the existing property rights and institutions that determine allocation can arrange for the efficient provision of these goods, i.e., whether they can secure the conservation of genetic diversity as an important component of global biodiversity. I reviewed empirical evidence on this issue and summarized the major findings as presented in the literature.

The empirical evidence indicates that the international agreements on the supply side of genetic resources mention both the *aim of conservation* and the *need for policy intervention*.

First, the emphasis on conservation implies that genetic resources are virtually considered as scarce resources. In the public discussion, the issue of scarcity is conveyed mainly by contrasting the indisputably economic values of genetic resources, as shown particularly in the context of agricultural breeding and world food supply, with the general evidence of biodiversity loss on all diversity levels. In scrutinizing these two aspects, it is typically concluded that genetic diversity is threatened by an irreversible reduction to suboptimal levels.

Second, the call for policy intervention implies that it is believed that the existing mechanisms for allocating genetic resources, i.e., the decentralized worldwide market allocation and the international but noncooperative allocation by administration, in their current design cannot arrange the efficient supply of genetic diversity. More specifically, while anecdotal evidence implies that the current situation in preserving genetic diversity is already suboptimal, it will continue to deteriorate if the appropriate measures to decelerate and halt the trend of decline are not enforced.

I described how the failure of an efficient supply of genetic diversity is connected to *property rights* and the question of the distribution of the economic rent associated with genetic information. Major conceptual findings in the literature were reproduced: most importantly, the market allocation of genetic resources can only provide a suboptimal supply. Suboptimality is attributed to the complex nature of the economic goods involved. Important factors are, on the one hand, the dependence of the resources' value upon the information embodied and, on the other hand, the specific requirements of a property right system for information goods. The conflicts in the existing system complicate the design of property rights that are needed to create incentives for effective in situ conservation. A solution to this problem is not apparent.

In this respect, it turned out that a market failure does not necessarily prevail because of a fundamental lack of property rights that renders genetic material an open access resource. Empirical evidence on bilateral transactions with genetic resources also indicates that the market does not fail in all respects. Moreover, given the different forms of genetic resources, the market may fail for specific forms of genetic resources, while it functions quite well for others.

In this context, the literature also argues that existing markets do not seem to internalize the potential use value of genetic diversity for future generations. For this reason, market values in a pure market allocation understate the scarcity of genetic resources compared to other goods. In other words, the market price is typically below the actual social value of genetic resources.

Commercial Use of Genetic Resources as an Effective Means to Conservation?

The other major question that I have investigated in detail in this chapter is whether the interest in the commercial use of genetic resources can induce private efforts towards the conservation of in situ biodiversity. This relates to the *market-based approach* to conservation. For this strategy to be effective (1) the (private) economic actors responsible for resource management must be able to appropriate sufficient parts of the economic value of genetic information and (2) the value itself must be large enough to compensate for foregone profits generated from land use other than conservation.

I examined the impact of trade in genetic resources upon the preservation of the world's gene pool, as well as upon the conservation of biodiversity habitats more generally. Regarding the criteria for analyzing and evaluating environmental policy instruments, this question refers to the *environmental effectiveness* of market-based conservation.

As with the question of an efficient supply of genetic diversity, the question regarding the appropriation and distribution of the economic rent is connected to *property rights*. Consequently, my analysis drew upon the previous description in this regard. While the *CBD* representing the predominant regime in biodiversity policy supports the market-based approach to conservation, the connection between the commercial use of and demand for genetic resources from in situ habitats as a vehicle for conservation seems less pronounced for *PGRFA*. I demonstrated that as far as this linkage exists, it is not primarily addressed within a decentralized market mechanism, but rather according to a state-centered approach. This result was supported with empirical evidence on the extensive use of ex situ materials in breeding processes.

The analysis of trade in genetic resources is embedded in a multistage framework of interdependent economic institutions. In this respect, environmental effectiveness of the market could be impeded by the imperfect design of its *institutional environment* and potential *policy failures* leading to price distortions. Both shortcomings favor the nonsustainable use of biodiversity in the end. I did not analyze these issues in detail, since they are addressed extensively in other social sciences.

In addition, *transaction costs* for the participants in the market for genetic resources can affect the extent of market transactions and, therefore, influence environmental effectiveness. I briefly reviewed the literature on transaction costs in the context of natural resources and related the findings to the trade in genetic resources. I demonstrated that transaction costs are partly addressed on a conceptual level in the studies on property rights. From an empirical perspective, so far there is only anecdotal evidence on this issue. I concluded that if the transaction costs are indeed large and impede the functioning of the market, the extent to which the existing institutional framework of the market is responsible for the extent of the transaction costs has to be studied. Alternatively, political conflicts between the stakeholders may lead to distortions in this regard. This question refers back to the issue of potential imperfections in the institutional environment.

Without neglecting the importance of these factors, I pursued a conventional neoclassical approach. More precisely, I assumed that no further institutional frictions exist and only relative prices drive the allocation of genetic resources and natural areas as their in situ habitats. I justified the focus of the analysis for the following reasons: it is frequently hypothesized that—independent of a proper design for the institutional environment and the recognition of property rights to genetic resources—resource providers in most cases may not reap substantial financial gains that they could, in turn, reinvest in preservation. This is because from the perspective of private stakeholders, genetic resources virtually do not represent scarce resources relative to other goods that are produced by land use other than conservation.

Given that property rights to genetic material are well-defined and enforceable, relative scarcity is ideally expressed in the market price for genetic resources. In order to evaluate environmental effectiveness, I identified and assessed empirical data on *market prices* and connected *quantities* and implicitly compared the data to potential revenues from alternative and more extensive land use.

For various reasons, in particular because of the heterogeneity of the genetic resources in the different sectoral uses, as well as the frequently confidential handling of transactions with genetic resources, it was hardly possible to draw a complete picture of the market. Accordingly, the figures derived are rather insufficient to either reject or accept the null hypothesis, indicating that the described market-based strategy to conservation is not effective as a whole.

In more detail, the empirical information summarized in this chapter suggests that, in individual cases, the trade in genetic resources for *conventional use* may generate incentives for effective conservation. This seems to apply to the use of wildly growing medicinal plants used for the production of botanical medicine. The impact of these uses on conservation depends finally upon whether a management regime for sustainable resource extraction can be implemented and effectively controlled. However, the global size of the areas preserved for this purpose could not be identified. Suggestive evidence was found for certain market impacts induced by local markets in the developing countries. In total, sectors with conventional uses display a smaller world market size than sectors with predominately informational uses. However, this difference in market size provides little evidence on the total contribution of natural genetic information to the value added of final products in the various sectors.

Considering the *informational use* of genetic resources, it turned out that data on market quantities and prices are often lacking. The empirical evidence presented suggests that just because of the nonrival use of genetic information, combined with the ability to replicate the information in ex situ conditions, there is often a low demand for specific in situ material, even if it has proven to be promising.

Furthermore, the literature indicates that a low individual willingness to pay for a specific genetic resource is also driven by *uncertainty* with regard to the quality of the embodied genetic information and by the experienced low probabilities of developing a new pharmaceutical or crop variety.

In addition, genetic resources as individuals of the same species carrying identical genetic information often show a wide geographical spread. If in situ genetic resources are supplied at low marginal cost, many suppliers of the same or similar species compete with each other. In this respect, the literature assumes that the commercial users possess a certain *market power* and force providers to operate at marginal cost.

The combination of nonrivalry, uncertainty, and supply side competition suggests that the market price for genetic resources is in general comparatively low. This in turn supports the assumption that—in the ordinary case—in situ providers of genetic resources have to expect relatively few market revenues. Consequently, in many cases alternative land use leading to a depletion of biodiversity is apparently the preferred choice.

Furthermore, markets for genetic resources can hardly induce *long-term conservation* if promising genetic information from specific sites can be stored and replicated in ex situ conditions. In this case, the commercial users of genetic resources who have contracted landowners at a specific site at some time may induce short-term conservation, which may, however, not prevail in the long run.

Regarding informational use in several sectors in detail, I illustrated that transactions of genetic resources for agricultural *plant breeding* are currently organized in ways that do not assign market prices to genetic resources and do not directly channel revenues to the in situ suppliers. In contrast, commercial users in the *pharmaceutical industry* typically purchase genetic resources upon the basis of bilateral market-like arrangements. I provided a summary of anecdotal evidence on transactions with genetic resources that are arranged in bioprospecting contracts and that are subject to access and benefit sharing provisions.

In addition, I reviewed *economic studies* that describe the commercial use and trade in an *analytical framework* and the derived numerical figures for private resource values. The focus of these studies falls upon determining the private willingness to pay for access to genetic resources. By (implicitly) comparing the derived values with land values for land use other than nature preservation, i.e., by considering a partial equilibrium model for a land market, general implications for the impact of the trade in genetic resources are derived. Since market-like transactions are primarily observed for resource use in the pharmaceutical sector, this industry is the focus of the literature.

The study that has recently created a major stimulus is the study by *Simpson et al.* (1996). I therefore placed it at the center of my analysis. I studied how the land value in the SSR model comes about and how these values change when (1) more recent empirical data is used as an input in the numerical simulation and when (2) alternative but plausible assumptions regarding the characteristics of genetic resources and their commercial use are made. Furthermore, I discussed how the economic, technological, and biological environment of the commercial use of genetic resources influences the private resource value. Tables 8 and 10 provide a qualitative summary as to how the alternative data and economic assumptions influence the resource value in comparison to the original numerical results. Furthermore, the impact of alternative assumptions regarding the ecological environment was discussed.

My study basically confirmed recent appraisals in the literature saying that, so far, the use of analytical models and simulations cannot determine the private value of genetic resources with sufficient accuracy. On the one hand, it seems to be unavoidable to make simplifying assumptions regarding the R&D process in theoretical models to keep the analysis tractable; on the other hand, it is because of these very simplifying assumptions in connection with difficulties in the choice of appropriate parameter values for numerical simulations that the (numerical) results derived from these models should be considered with care.

A general caveat to a market-based strategy on conservation is that the market price of genetic resources may fluctuate over time. Consequently, even if the market does effectively create incentives for conservation, varying prices induce varying revenues for the landowners. When market prices temporarily decrease, landowners may respond with the conversion of their natural areas. Whenever these areas represent habitats of biodiversity that is sensitive to habitat change, restoration may not be possible when market prices increase. Consequently, in the presence of ecological thresholds, markets fail to guarantee biodiversity conservation. Accordingly, market allocation needs to be supplemented with a regulatory approach to resource use. The following chapter provides a detailed analysis of this approach to conservation, which is considered as complementary to the market-based approach described.

4 Preserving Biodiversity as a Global Public Good: Protected Areas and International Transfers

4.1 The Economics of Protected Area Policies

The previous chapter investigated how the demand for private goods of biodiversity can jointly contribute to the safeguarding of public good-like ecosystem services. In the following, I study instruments and arrangements that directly address the *public goods of biodiversity*. In this context, I assume that human *land use* essentially influences and determines the extent of biodiversity and size of ecosystem services. Accordingly, a major instrument to secure the supply of public goods of biodiversity is the designation of *protected areas*, i.e., natural areas that remain close to their natural state and undisturbed by human use (van Kooten and Bulte 2000: 311). Frequently, the exclusion and control of human disturbance is vital for the ongoing provision of sensitive but valuable ecosystem services.

In this chapter, I analyze the conceptual basis of protected area policies (Section 4.1.1) and its current outcome (Section 4.1.2). In doing so, I place the focus upon the international aspects. In the remainder of the chapter, I conduct an empirical analysis of the Global Environment Facility (GEF), which serves as an international transfer mechanism that, among other things, assists protected area policies in the developing world (sections 4.2–4.4). The results are summarized in Section 4.5.

4.1.1 Positive Externalities and Biodiversity as a Global Public Good

Policymakers consider protected areas an appropriate instrument when two properties are satisfied. First, preserved natural sites generate *positive externalities* that are not captured on a market or ecosystem services generated from those sites display the nature of a *public good*. Second, natural sites represent productive land that is relatively scarce, i.e., there are *competing uses* for biodiverse natural sites (see Section 2.1). In a case where conservation generates positive externalities or supports the provision of public goods, (private) landowners do not obtain sufficient returns to preserve and manage ecosystems to a socially optimal extent. Moreover, given alternative profitable land use, the ecosystem is modified or converted, with the consequence that the flow of socially valuable ecosystem services is reduced to below optimal levels. Thus, positive externalities are not internalized and public goods are undersupplied. In order to prevent such ecosystem changes, policy interventions are needed.

Economic analysis broadly uses the concepts of externalities and public goods. A variety of definitions are used for them. While externalities refer to costs and benefits of certain activities, public goods are described by their nonrival and nonexclusive consumption. Cornes and Sandler (1996: 6f.) argue that it is helpful to view externalities and public goods as *incentive structures* instead of relating them inherently to specific actions. In this regard, externalities are considered as a family of market failures, with public goods being a member thereof. Externalities are the basic concept to define a problem. In order to describe it in detail, additional structure is imposed.

Several public goods exist in the context of biodiversity (some of which have been described in Figure 1 in Section 2.1.1):

- *Global public goods* are ecosystem services that generate benefits for more than a group of countries and a broad spectrum of the population. They are related to the existence of species and ecosystems and carbon sequestration (OECD 2004: 36). Because carbon sequestration helps mitigate the impact of climate change, it represents an intermediate public good that supports other supporting ecosystem services. In this respect, carbon sequestration generates indirect use values. The existence of biodiversity contributes to cultural services and possible future provisioning services. It provides option and nonuse values.
- *Local public goods* are ecosystem services, such as flood and erosion control. The literature classifies these services as regulatory services. They contribute to indirect use values and, by definition, generate benefits that occur more exclusively on a local level. While the use of these services by one party does, in many cases, not diminish the quantity available to the others, access to the services is sometimes made exclusive (club good). In the same way, the public sector provides some ecosystem services that are rival in use, nonexclusively (genetic material for breeding in agriculture). Consequently, ecosystem services sometimes do not exist in their original form but represent social constructs determined by human-devised institutions (Kaul and Mendoza 2003).

The public good properties of certain ecosystem services raise questions as to (1) who should provide the good "biodiversity" and (2) how the costs of its provision should be shared. When governmental authorities typically take responsibility for implementing a mechanism for public good provision, a further question concerns (3) how the decision makers can obtain the necessary information (Heal 1999b).

Returning to protected areas as of means of preserving specific ecosystem services, the *Convention on Biological Diversity (CBD)* defines a protected area as "a geographically defined area, which is designated or regulated and managed to achieve specific conservation objectives." These objectives are addressed by policy instruments that (1) indirectly control human ecosystem use and support the internalization of the values of ecosystem services in private land use decisions (see Section 2.3.3) or (2) directly prohibit or restrict certain ecosystem uses.[147]

Regarding indirect interventions, which typically address the private costs of ecosystem use or the income of the resource users (Perrings and Opschoor 1994), the impact of protection is subject to fluctuations in the input and output prices and landowners' income, which both serve as the determination base targeted by the intervention. For sensitive ecosystems, however, such variances in the induced level of conservation can lead to disturbances that trigger irreversible undesirable ecosystem changes. For this reason, direct interventions in the form of a combination of command and control instruments and quantity-based instruments are the preferred tools. The restriction of ecosystem use in this regard is consistent with the concepts of the *safe minimum standard* and the *precautionary principle* in environmental policy (e.g., Bishop 1978; Tacconi and Bennett 1995; see also Section 2.4).[148] Consequently, in this respect, there is good reason for protected areas, even if ecosystem services can be brought onto the market.

Depending upon the specific protection needs and the regulatory approach to ecosystem use, a spectrum of human use that leaves the ecosystem close to its natural state is conceivable. As a generalization, let us perceive landscapes as a mixture of two parallel existing land use regimes: the protected area system and the system of relatively unregulated but intensive land use. Each of the two systems may itself be of a quite heterogeneous nature, possibly with only slight differences between neighboring uses.

Given the necessity of an intervention in ecosystem use, the protected area policy essentially faces two tasks:

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147 Under ideal conditions, internalization may not demand policy intervention but may occur spontaneously among private stakeholders: the private beneficiaries of non-market ecosystem services compensate the private landowners for foregone profit if they forgo such alternative uses (Coasian bargaining solution).

148 For a distinctive perspective on thresholds and command and control instruments, see Perrings and Pearce (1994).

- the definition of the *objective* of protection, i.e., where should protected areas be established and which uses should be restricted/allowed, and
- the *enforcement* and monitoring of protection.

These two tasks can be studied from both a *domestic* and *international* perspective. While the analysis focuses upon the international aspects of the enforcement issue in the following, I first briefly summarize aspects of selecting objectives of protection and enforcing them in the domestic context.

4.1.1.1 Policy Objective: Selecting What to Preserve

The formulation of objectives in protected area policies is an interdisciplinary task. While ecology is the leading discipline in this regard, essential contributions are made by economics and other social sciences. Conservation planning usually follows an integrated approach. In order to describe the disciplinary challenges to this process, I roughly distinguish the ecological dimension and the socioeconomic dimension.

Regarding the *ecological* dimension, certain difficulties in conservation planning arise due to problems in finding the right measure and indicator for biodiversity, as well as in identifying and describing its dynamic nature (see sections 2.1.1 and 2.3.1). Considering the strategic approaches to conservation, the debates in ecology surround two concepts: the *hot spot* approach focuses more narrowly upon the preservation of the habitats with a high species density. Since many biodiversity components may not be included in this concept, a more broadly defined approach of a *representative reserve system* is alternatively called for. A drawback of both approaches to conservation is that they rely upon the assumption of rather static ecological patterns (Armsworth et al. 2004; SCBD 2004).[149]

Further complexities in both strategic planning and operational enforcement arise due to an insufficient knowledge of the connectedness of ecological processes at different spatial scales, the inherent ecosystem dynamics, and the interplay between spatial and temporal variations. Additional uncertainty is caused by exogenous shocks, which can disturb preserved ecosystems. In spite of these difficulties, the literature has defined stages of a rational process of conservation planning and introduced principles, such as that of interconnectivity among conserved habitats, in order to mitigate impacts from unforeseen ecosystem changes (Margules and Pressey 2000; SCBD 2004).

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149 A related but ecologically and socioeconomically more integrated framework concept is the *ecosystem approach*, which pays particular attention to the maintenance of supportive ecosystem services (SCBD 2004; WRI 2000).

The *socioeconomic* dimension of conservation planning addresses the issues of efficiency and equity. By drawing upon ecological data, efficiency primarily refers to the cost-effective use of economic resources in the enforcement of conservation. The studies on *reserve selection* investigate these aspects (Ando 1998; Costello and Polasky 2004; Polasky et al. 2001). Reserves in this regard are defined as strictly protected areas (Margules and Pressey 2000). The literature studies how policymakers should reasonably make their choice between several candidate sites in order to obtain the maximum cost-effectiveness in conservation.[150]

In formulating the problem of reserve selection, the studies assume given figures on benefit and/or costs. They neglect the question as to who benefits and who incurs the costs of protection. However, the distribution is of concern, since particularly the cost sharing determines political acceptance and influences the enforcement and effectiveness of protection. Regarding the practice of conservation planning in developing countries, in many cases, the poorer population depends upon resource use in the ecosystems considered for protection. Equity aspects are concerned where local stakeholders have so far been unable to represent their interests in political processes. To establish equity, a *participatory approach* to the design and implementation of protected areas is needed. In order to reconcile diverging interests in the use and conservation of ecosystems, protection measures are frequently embedded in *integrated conservation and development projects* (ICDPs). The core element of these programs is the integration of conservation on the one hand and sustainable local use on the other.

This approach acknowledges that protected areas can serve several purposes in addition to conservation (Brandon and Wells 1992; SCBD 2004: 92). However, the effectiveness of ICDPs is still subject to debate (Ferraro and Kiss 2002). It is often questioned in the literature as to whether local ecosystem use can be reconciled with the conservation needs of a specific site. Finally, the choice of the appropriate system of protection seems to depend upon a combination of several factors that prevail at a specific site. These factors include the pressure from alternative land use, i.e., the economic costs of stricter protection or the compatibility of traditional and communal resources use with biodiversity conservation.

4.1.1.2 Domestic Policy on Protected Areas

Given that it is known what is ideally preserved, the target level of protection usually cannot be obtained without policy intervention because of the *positive*

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[150] While these studies usually consider the selection from alternative habitats, Weitzman (1998) and Metrick and Weitzman (1998) formulate the choice problem as a choice between alternative species.

externalities of biodiversity conservation or because of the *public good* nature of many ecosystem services (see sections 2.1 and 2.3.3).

The solution to the problems of internalization and public good provision is connected to the *property rights* that are assigned to land as natural habitat and the biological resources hosted therein. Generally, the absence of well-defined and enforceable property rights on natural habitats and resources defines them as open access resources, which suffer from degradation and overexploitation (Mendelsohn 1994). Therefore, the first step necessary in any protected area policy is to guarantee that an appropriate property rights regime is specified and enforced.

Although this requirement is often difficult to establish in practice, my analysis continues with the assumption that policies are not impeded in this regard. Figure 16 describes a way to classify regulatory approaches in domestic protected areas. For this purpose, private and communal property rights are differentiated from state property.

Considering first *state ownership* of natural areas, the property rights to biological resources hosted therein are usually tied to the rights to these areas. The responsibility for effectively managing the areas rests with the public authorities. In practice, the protection of these natural areas in the hand of the public sector manifests itself in a system of national parks and protected landscapes.

National parks generate ecotourism services as joint goods that, in turn, can be sold on the market. Consequently, protected areas can generate certain revenues. However, public national park entities hardly display profit-maximizing behavior and, in many cases, revenues from ecotourism do not suffice for the sustainable financing of the appropriate protected area management (see Section 4.2.4.1). Moreover, revenues from tourism add to the public funds the park administration is granted in order to cover the costs of the effective park management (LaPage 1994; James et al. 1999a).

Figure 16:
Regulatory Approaches in Protected Area Policy According to the Property Rights Regime

		Biological resources	
		State	Private/communal
Natural land	State	(Original) state ownership Land takings	Public private partnerships
	Private/ communal	Standards, charges, environmental taxes	Contracts, compensations

If market revenues can be obtained from managing publicly owned protected areas, they offer the chance for a *private public partnership*: private actors may be contracted for the management of such areas in return for tourism revenues. While the land property rights are left unchanged, the use rights to the land and biodiversity hosted therein are sold to the private sector (*Economist* 2003b). Donations from private foundations to public national parks represent another form of private public partnership (González-Montagut 2003). Here, the private donor may insist upon controlling the property rights to biological resources only to the extent that the effective park management is enforced. Finally, since (public) protected areas can host genetic resources that are promising for R&D, market returns may be obtained from access and benefit sharing arrangements between park authorities and commercial users (Laird et al. 2003; see also sections 3.1.2 and 2.3.2.2).

As far as natural areas are not originally in the hands of the public sector, the state can expand its holdings of protected areas by acquiring natural land from private landowners or local communities. For this, public authorities and landowners may bargain over the price for the land title. Alternatively, the authorities can exercise their power by law and take the land.[151] Through the use of such *takings*, the government can reject private property rights and substitute them with state property rights in order to preserve social ecosystem values. More frequently, however, the authorities compensate the previous private landowner. In contrast to bilateral bargaining, the government may unilaterally fix the amount of compensation (Kaplow and Shavell 1999; Innes 2000).

Turning to private/communal landownership, the close link between property rights to the land and the biological resources (species) hosted therein may be suspended by governmental regulations. More specifically, by implementing *standards* and *charges*, private property rights are restricted in choosing the profitable form of land use. In practice, such regulations concern the protection of endangered species on private land (Brown and Shogren 1998). As mentioned in Section 2.3.3, this sort of quantity-based land use regulation can be augmented to a cap and trade mechanism, i.e., it is combined with a market regime for transferable private land use rights (transferable development rights) (Panayotou 1994; Kulessa and Ringel 2003). An intervention into private property rights is also represented by a price-based regulation in the form of an *environmental tax*.

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[151] In the real world with incomplete property rights, the state may also expand its holdings by assuming property rights in areas that, so far, have been open access resources and for which, until recently, no alternative uses have prevailed.

In contrast, other price-based regulations that subsidize biodiversity-friendly land use leave private property rights untouched (Goeschl and Lin 2004).[152] Such payments to landowners address prices and costs in order to influence private land use decisions and make the landowners internalize positive externalities from undisturbed natural areas. When private landowners can choose between alternative land uses, subsidization effectively represents *compensation* for opting for the less profitable, but biodiversity-friendly, land use (Ferraro and Simpson 2002).

Regarding the arrangement of compensation payments, two general forms are conceivable: (1) payments can be defined conditionally on whether the private landowner commits himself to forgo land use that affects biodiversity negatively. In a specific case, the commitment to pay compensation can be fixed in a *contract* between private landowners and public authorities as the donors.[153] When the commitment to preserve still allows a landowner to generate private goods and obtain some market return for them, compensation payments are equivalent to a subsidy for natural areas as input into biodiversity-friendly production. Alternatively, (2) payments may not directly refer to natural areas but represent an output subsidy for biodiversity-friendly production or a subsidy for input other than land (Ferraro and Simpson 2002). An example of the latter is public capital transfers for infrastructure development in ecotourism.

Given the positive description of regulatory approaches in domestic protected area policies, a normative analysis of efficiency and effectiveness is reasonably the next step. For reasons of space, a detailed analysis of these aspects is not provided here. Major questions that the literature addresses in this regard are the following:

- Regarding the alternative state versus private land property rights, comparative analyses are carried out to identify the advantages and disadvantages of each regime (Kaplow and Shavell 1999).
- Given that policies on land takings can create perverse incentives for both the private landowner and the regulator, some studies ask how policies should be designed to mitigate such effects (Innes 2000; Innes et al. 1998).
- With respect to private land use, studies analyze under what conditions the joint production of public and private goods in protected landscapes supports biodiversity conservation to an efficient extent (Heal 2003; Holm-Mueller 1999).

[152] A specific form of price-based regulation is the removal of perverse subsidies (Goeschl and Lin 2004).

[153] Considering long-term protection, the financial means used for the compensation of private landowners may alternatively be directed to the acquisition of the land, i.e., the resource stock that provides the valuable ecosystem services.

- The choice of the optimal policy instruments for regulating private land use is another topic of investigation. It relates to the comparative analysis of market-based regulation compared to regulation by charges (Siebert 2005: 130f.). Given the nature of biodiversity, this question can be analyzed against the background of uncertainty (Weitzman 1974; Baumol and Oates 1988: 190ff.).
- Considering price and quantity-based regulation, studies ask for the optimal design of payments to compensate for private conservation (Muller and Albers 2004; Ferraro and Simpson 2002).
- Furthermore, considering tradable permits for land development as a specific type of a quantity-based instrument, analyses ask whether such a regime can generate cost savings and to what extent it is impeded by difficulties in defining a proper indicator for measuring and comparing biodiversity at different locations (Weber 2004).
- Regarding the extensive data needs and uncertainties in the regulatory process, the impact of imperfect information upon the outcome of this process is a further topic in the literature (Goeschl and Lin 2004; Polasky and Doremus 1998).

While these questions relate to the concept of environmental externalities and the question of the optimal policy instruments, the four scenarios depicted in Figure 16 can also be related to the question of public good provision.

In a state-centered allocation, the public sector responsible for the provision of the (pure) public good decides upon the quantity of the good. In this case, the taxpayers carry the costs of provision. When private property rights to land and/or biological resources are involved in the allocation, it is more difficult for the regulatory authorities to control the exact quantity of conservation as a public good. The effectiveness of regulatory policies depends upon how they can influence the private incentives to forgo land use that is harmful to biodiversity. Relative prices for private goods that are supplied jointly with public goods largely control these incentives. By imposing environmental taxes upon private land use, the costs of the public good provision are initially incurred by the landowners. However, landowners who sell agricultural or forestry products may be able to transfer a part of the costs to the consumers. When the state concludes contracts for private conservation, the taxpayers carry the costs. In the case of private public partnership, the benefit principle determines the cost sharing: those people on the demand side of ecosystem services who care the most about biodiversity and potentially derive the largest benefit from its conservation make contributions to its finance.

4.1.1.3 International Policy on Protected Areas

Positive externalities, or, synonymously, spillovers from protected biodiversity sites, do not stop at national borders. *Global spillovers* from biodiversity conservation represent the special case of cross-border ecosystem services that generate benefits for a broad spectrum of the global population, whereas the ecosystem owners in the resource countries are not compensated for the costs of provision. As discussed above, external benefits of conservation can virtually be nonrival in their consumption. Together with the inability to make them exclusive, they represent pure *global public goods* (Sandler 1993; Anand 2004).

Since resource countries cannot appropriate the external benefits generated from their conservation efforts, in particular from the management of the national protected area system, the cross-border spillovers from their efforts are systematically disregarded in their policy. Domestic investments in the protection of biodiversity within their own national boundaries are, at best, sufficient to attain a level of conservation that is optimal from a national view. Whether an effective domestic policy can also safeguard conservation that is optimal from a global perspective depends upon the *aggregation technology* underlying the provision of conservation as a global public good.

In general, four alternative concepts of technology are identified (Anand 2004; Sandler 2002). Suppose that there are many countries, each of which enforces measures contributing to an international/global public good. For n countries, let the contribution of a country, i, $i = 1..N$, be denoted by g_i; the overall level of the public good is G. In light of this, a *summation technology* assumes that the efforts of all countries contribute to the overall level of the global public good, $G = g_1 + ..g_i + ..g_n$. A *weighted sum technology* assumes a similar structure, $G = \omega_1 g_1 + ..\omega_i g_i + ..\omega_n g_n$. The difference is that individual efforts are not perfectly substitutable. The *best-shot technology* assumes that the level of the public good is determined by the largest individual effort, $G = \max[g_1,..g_i,..g_n]$. In the case of a *weakest link technology*, the provision of the good crucially depends upon the participation of all countries and is determined by the smallest individual effort, $G = \min[g_1,..g_i,..g_n]$. For a summation or weighted sum technology, it turns out that the level of the pure public good in the noncoordinated outcome falls short of the optimal level. In contrast, for a best shot and weakest link, an efficient supply may be attained under specific circumstances (for a discussion, see Sandler 2002).

Regarding biodiversity of global importance and the ecosystem services representing pure public goods on an international level, a weakest link technology only applies if biomes are interdependent in such a way that each biome is extremely sensitive to changes in the neighboring biomes. In contrast, the transboundary values generated solely from the existence of species and the redundancy of organisms of the same species may imply that a best-shot technology can describe biodiversity conservation. Nevertheless, as discussed in Chapter 2, the problem of conservation cannot reasonably be reduced to the safeguarding of endangered species. Moreover, the dynamic nature of biodiversity and the interactions between the different levels of diversity have to be taken into account. Accordingly, a *summation* or *weighted sum technology* seems to be appropriate in order to describe the provision of biodiversity as a global/international public good. Therefore, I conclude that although domestic policies on national protected areas generate international public goods as joint products, these goods are undersupplied. The reason for this result is a *market failure* for public-good-like ecosystem services of global importance.

In the following, I recapitulate the theoretical foundations for internalizing biodiversity spillovers on an international level and providing biodiversity as an international public good. In order to conceptualize the international policy in this regard, I assume that domestic ecosystem use by private firms and households is controlled by governmental decisions, i.e., on an international level, each country is regarded a uniform actor in the decision-making process. In contrast to the domestic level, countries are not subject to a regulatory authority that can force them to comply with certain general conservation objectives and forgo certain ecosystem uses if necessary. The absence of a supranational authority means that the *sovereign countries* possess the property rights to the natural resources within their territory. For this reason, internalization on an international/global level demands the voluntary *cooperation* of the countries involved (Ferroni and Mody 2002).

Unidirectional and Multidirectional Spillovers and Mechanisms for Internalization

Cooperation in this context requires it to be profitable for a country to cooperate. In other words, each country experiences a net increase in well-being if it becomes a member of an agreement and fulfills its membership obligations (given that all other member countries fulfill theirs). Furthermore, since property rights to natural resources are assigned to the countries, cooperation has to include a mechanism for the exchange of property rights. The design of this mechanism depends most importantly upon the *direction of spillovers*. More precisely, it needs to be known (1) which unilateral actions by whom induce spillovers/public goods and (2) who benefits from them. Given that biodiversity is a fundamental property of ecosystems worldwide, there is in fact plenty of evidence of externalities that are unidirectional or multidirectional.

Regarding the case of strong *multidirectional* spillovers, each country benefits from the efforts that every country makes with respect to the protection of their biodiversity endowment. In order to make national governments take into account the spillovers of their policies, an agreement on a reciprocal increase of national efforts would lead to an increase in the overall level of conservation and a Pareto improvement of the participating countries. Generally, an agreement of this kind, which is based upon *reciprocal physical efforts*, does not need to include international financial flows. The arrangement can ensure that each country enforcing physical measures within its own territory also carries the incurred costs alone (Endres 1995).

Unidirectional spillovers result from the uneven allocation of biodiversity among countries. The relevant spillovers, i.e., public-good-like ecosystem services are generated from resource management in the biodiversity-abundant countries. Resource management in the less abundant countries is only of minor importance to the remaining countries. Accordingly, an agreement on international cooperation provides for additional protection activities in the resource-abundant countries only. The remaining countries are, in turn, obliged to participate in the financing of the protection activities that generate the international spillovers. In other words, the countries receiving spillovers/public goods provide *compensation* for the resource-abundant countries to cover the costs of protection in excess of the domestically optimal level (Endres 1995; Cervigni 1998).

When considering empirical examples of cross-border biodiversity spillovers and the mechanisms for internalization, elements of reciprocal efforts and compensation in international agreements can be found. For instance, *river pollution* between upstream and downstream users represents a classic example of *unilateral* environmental externalities. In the case of cross-border externalities, internalization is addressed through an agreement between the governments of the countries involved. As shown, for example, by the case of chloride pollution in the Rhine River, Germany and the Netherlands have provided compensation to France as the upstream country to introduce facilities to reduce pollution (Barrett 2003: 128ff.; Ströbele 1991).

International *multidirectional* spillovers are present when, for example, the functioning of ecosystems located within different countries depends upon the ecological integrity of the neighboring ecosystems. Ecological linkages of this kind become visible, inter alia, in the migration of species across national borders. A way to ensure internalization in this context can be an agreement on a regional or continental level that commits the participating countries to establishing a sufficient national protected area system. In the European Union (EU), an approach to reciprocal conservation efforts is agreed upon in the *Habitats Directive* on the Conservation of Natural Habitats and Wild Fauna and Flora (92/43/ECC), which aims at the establishment of a cross-border network of protected ecosystems (*Natura 2000*). Consequently, this directive arranges the reciprocal efforts of the EU member states. In addition, some of the several mechanisms within the Union providing transfers for various purposes directly or indirectly offer funds for the conservation of European biodiversity. The *LIFE* mechanism in particular aims at the funding of conservation projects in accordance with the Habitats Directive with its LIFE-Nature program (EC/EDG 2003).[154]

A further example of international multidirectional biodiversity spillovers refers to ecosystem use in the *Antarctic*. In 1991, the 26 consultative parties of the 1959 Antarctic Treaty that claim property rights to the natural resources agreed upon a reciprocal *ban on mineral exploitation* for the next 50 years in order to safeguard the continent's vulnerable ecosystems (Cullen 1994; Barrett 2003: 117f., 156f.).

Regarding *global spillovers* from biodiversity conservation, the *CBD* is the major international environmental agreement in this respect. Its provisions apparently address both multidirectional and unidirectional externalities: CBD Art. 8 calls for the reciprocal efforts of the CBD signatory countries to establish *national protected area systems*. Nevertheless, in CBD Art. 20.2 the developed countries are called upon to assist the resource-abundant developing countries in their efforts to conserve the globally important biodiversity residing on their territory. More specifically, developed countries are called upon to provide *transfers* ("financial resources") that are invested in the conservation and sustainable use of biodiversity in the developing world. This means sustainable resource management in the developing countries generates global public goods whose provision is to be financed by the developed countries. Without ignoring developed countries with substantial endowments of biodiversity, developing countries typically represent the resource-abundant countries.

The public discussion implies that the unidirectional spillover from developing countries to the developed world (and the remaining developing countries) is regarded as the dominant type of global spillover. For this reason, political efforts and academic analysis focus upon the issue of international transfers for the conservation and financing of biodiversity as a global public good (Sandler 1993; Perrings and Gadgil 2003).

Incentive Problems in International Biodiversity Policy: Efficiency-Equity Interplay

The fact that the agreements on transfers and additional conservation, and on providing funding for the global public good, must be reached on a voluntary basis implies that the *participation constraint* of each of the countries involved

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[154] LIFE means *L'Instrument Financier pour l'Environnement*.

must be satisfied. In economic terms, each country aims at the maximization of its own payoff from cooperation. These payoffs are fed by the surplus created from the reallocation of resources resulting from the cooperation between countries. In this context, each country may display opportunistic behavior or act strategically in order to maximize its share of the surplus.

Opportunistic or strategic behavior evokes cases where an agreement between countries may not be reached or where the agreement cannot establish a Pareto optimal outcome. In case of global biodiversity, the quest for the acceptable distribution of the surplus generated by cooperation aimed at the protection of biodiversity influences the level of conservation and, thus, the size of the surplus attained (Cervigni 1998; Mohr 1990; Sandler 1993). In this regard, the *interplay between efficiency and equity* influences the effectiveness of international biodiversity policy (see Section 4.3).

Suboptimal outcomes in international policy can be attributed to different forms of *incentive problems*. In order to describe them, let us assume that resource countries, or, synonymously, developing countries, negotiate with developed countries, which represent the transfer donors. Both parties bargain about the level of resource conservation in excess of the present domestic level and the size of the international transfer that the donors provide in return. Through the use of the transfer, the donors attempt to influence the management decisions of the sovereign resource countries. They may grant untied transfers or, more commonly, provide them on a conditional basis for conservation actions agreed upon beforehand.

Regarding the number of resource and donor countries involved, various scenarios are conceivable. For example, (1) conservation in one resource country generates global spillovers and all remaining countries form a donor community that negotiates with the resource country concerning safeguarding the flow of spillovers (Cervigni 1998). Alternatively, (2) the same spillovers may be generated by several resource countries, thus creating competition between them to receive payments by the donor community (Stähler 1992, 1994). Finally, (3) certain spillovers are only experienced by some (developed) countries. In this case, the resource country and the donor country bargain face to face on a bilateral basis.

All of this can lead to the following incentive problems:

- Dumping strategies of resource countries in the context of competition for international transfers given irreversible resource degradation.
- Strategic noncooperative behavior of resource countries in the short term in order to attain a larger share of the cooperation surplus in the long term.
- Opportunistic behavior of resource countries due to asymmetric information concerning the costs and other determinants of conservation.

- Free-riding behavior among donor countries in mobilizing funds needed to compensate resource countries for conservation on behalf of the donor community.

The *dumping strategy* is only viable for a resource country if many countries compete for transfer payments by the donor community, and the ongoing flow of compensation for the maintenance of globally important ecosystem services that donors provide is relatively inelastic in its size. Furthermore, the strategy requires these ecosystem services to be supplied initially by several resource countries. Since the supply is not costless, any country that is not compensated abandons conservation, with the consequence that these ecosystem services cease indefinitely within the country in question. Consequently, the number of suppliers decreases and fewer countries compete for compensation in the subsequent periods (Stähler 1992, 1994).

In the competition for transfers, a resource country has an incentive to demand compensation below the conservation costs and, thereby, to attract a large portion of the transfer payments. At the same time, this forces competitors to abandon conservation and drop out of the group of future providers. Consequently, the dumping country increases its bargaining power in the long run. In future negotiations, it can demand compensation above the conservation costs and, thus, appropriate a positive conservation rent (Stähler 1992, 1994). Given that the donors provide a flow of transfers whose size remains relatively constant over time and act as price taker, the extent of conservation depends upon the compensation demanded by the dumping countries. The resulting path of conservation is characterized by a relatively high level in the short and medium term but by a suboptimally low level in the long run.

In order to limit the profitability of a dumping strategy and, thus, mitigate the distortions of the allocation, the donors may use a transfer scheme that enables a minimum number of resource countries to be contracted for conservation in each period, independently of the favorable conservation price certain resource countries indicate. As a consequence, it would be difficult for any of them to obtain excessive bargaining power in the long run (Stähler 1992, 1994).

Irreversibility in biodiversity conservation also offers resource countries the chance to act strategically and *refuse cooperation for strategic reasons*, at least in the short or medium term. The idea is that in the negotiations with a resource country, the donors initially make an offer regarding the size of the transfer and resulting level of conservation in that country. Both parameters determine the division of the surplus of cooperation between the two sides. Because the level of conservation is subject to the sovereign decisions of the resource country, it is at liberty to reject the offer. Moreover, it may credibly threaten to deliberately deplete its own resources ("Burn the forest!"), if the donors do not increase their transfer offer and thereby change the distribution of the cooperation surplus to its favor (Mohr 1990). Consequently, the irreversibility property improves the bargaining position of the resource country. The donors are called upon to relinquish parts of their share of the cooperation surplus in order to ensure conservation.

If the donors do not react and the resource country carries out its threat, i.e., it depletes its natural resources, the biodiversity spillovers that the donors receive will cease. When countries repeatedly interact and the donors constantly refuse to suggest an alternative distribution of the cooperation surplus, biodiversity and, with it, the size of the attainable surplus shrinks from period to period (Sandler 1993).

In order to avoid such a scenario, it is important that donor countries are aware of their bargaining position in the negotiations. They should attempt to reach an agreement at an early stage in order to safeguard an adequate level of conservation and change the incentives of the resource country so that depletion is no longer a credible option (Sandler 1993).[155]

In addition to strategic interactions in the presence of irreversibility, the interaction between resource and donor countries can be affected by problems of *asymmetric information* (Kölle 1995): first, prior to an agreement, the donors may not know the true costs that accrue to the resource country when it enforces the agreed upon level of conservation (hidden information). Second, the donors may not observe precisely whether the final level of conservation is attributed to exogenous factors, for example, specific climatic or ecological incidences, or whether the resource country has indeed carried out the measures for which it is compensated (hidden action).

Given these information asymmetries, a resource country has an incentive to overstate the actual costs of conservation and/or refrain from the agreed upon actions. By not telling the truth or by noncompliance, the resource country manages to appropriate an information rent as a portion of the cooperation surplus. From the perspective of the donors, this results in an excessive relative price for conservation. They therefore also provide a comparatively smaller amount for transfers. Accordingly, the agreed upon level of conservation and size of the surplus fall short of those in a situation with perfect information. Nevertheless, when applying findings from the standard model of the agency theory with asymmetric information (Varian 1992: 440ff.) to this situation, the donor countries anticipate the opportunistic behavior of the resource country and offer payment schemes that take the asymmetric allocation of information into account. These schemes aim at the incentives of the resource countries to reveal the true cost of conservation and to comply with the measures agreed. The

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155 Several factors, like the costs of depleting the resource stocks and the potential domestic benefits from resource conservation, have to be considered, since they determine whether the threat is credible anyway.

literature shows that although such incentive-compatible payments mitigate the impact of the asymmetries, it is not possible to attain an optimal level of conservation in this way (Farell 1991; Buchholz and Haslbeck 1991).

Aside from incentive compatible payments, associated signaling activities are available to reduce information asymmetries (Varian 1992: 440ff.). Donors may employ these instruments in the context of international transfers for conservation actions, although their employment can be quite challenging in practice (Kölle 1995). Furthermore, donors and resource countries may reduce information asymmetries by collaborating on the process of planning and enforcing the conservation actions for which transfer payments are requested.

When many countries benefit from unidirectional biodiversity spillovers, *free riding among donors* represents a major incentive problem. When several countries benefit from cross-border ecosystem services, this implies that these services are nonrival in their use. Examples are the existence or bequest values of biodiversity that are appropriated on a global level. Since these values are usually nonexclusive, the ecosystem services represent *pure global public goods*. In order to ensure their provision in the biodiversity-abundant resource countries, the beneficiary countries (1) may provide funding unilaterally and in a noncooperative way or they (2) may cooperate and arrange for a financing mechanism that is implemented for this purpose.

Two factors impede cooperation between the beneficiary countries (or, alternatively, donor countries): first, every country is sovereign in its decision, not only with respect to the use of its own natural resources but also with respect to its participation in international agreements. Second, each country has private information concerning its true preferences for the ecosystem services that represent the global public goods. These two factors are consistent with the two forms of free friding behavior. *Free riders* are

- countries that do not join an agreement that arranges for the financing of transfers for conservation although these countries receive benefits from internationally assisted conservation measures, and
- countries that actually contribute money for transfers but only to a small extent that does not represent the benefits they actually receive.

Imperfections in these two respects cause the failure of the donor community to provide enough funds to compensate resource countries for conservation that reaches a globally optimal level. Given this result, empirical evidence on environmental agreements presented in Section 2.2 suggests that, in practice, the incentive of nonparticipation is not dominant. The literature in this regard analyzes whether and how first-best or second-best outcomes of international cooperation can be supported. It turns out the *nature of the international environmental problems*, i.e., the number of countries involved, the allocation of costs and benefits, and/or the *design of the negotiations*, has an influence on the attainable outcome. The tools of public economics and game theory have been applied to study these issues (Sandler 1993; Sandler and Hartley 2001; Barrett 2003: 195ff.; Carraro and Siniscalco 1993, 1998). Section 4.4 readdresses some of these questions in detail.

Cooperation among Donors Versus Unilateral Actions

Donors may cooperate and conclude an international agreement concerning the obligation to finance conservation in the developing world. This represents a *multilateral* approach to preserving global biodiversity. Alternatively, a donor country may provide funds for conservation abroad on a *unilateral* basis. This applies to a case where coordination among countries fails and no donor agreement is reached. Furthermore, an individual country may provide unilateral funding for third countries in order to safeguard the local biodiversity endowments that generate spillovers that more exclusively accrue to the individual donor (impure global public goods). Finally, a country may provide unilateral funding in addition to its commitment to a present cooperation. This is because the country either feels that the funds the donor community collectively mobilizes are insufficient to conserve the components of global biodiversity that the country itself considers particularly important or that it simply wants to encourage other donors to increase their funding in the future (Hoel 1991).

Economic studies investigate whether *unilateral*, or synonymously, noncooperative, funding on a *conceptual basis* represents an effective instrument to mitigate global environmental problems including global biodiversity loss. In the classical noncooperative setting of a private public good provision, it turns out that the unilateral contributions are below optimal levels, since each donor country ignores the external benefits of its contribution to the other countries. Accordingly, the size of total funds and, therefore, the level of induced protection of biodiversity fall short of the optimum (Barrett 1994a, 1994b).

Several other studies assume a modified setting for the private public good provision game. For example, Hoel (1991) considers both noncooperative interaction and cooperation but uses the term unilateral action in a different sense: he defines it as the contribution a donor country makes in excess of what its payoff function dictates. In other words, the donor country acts altruistically rather than selfishly in that it intends to "set a good example." The results of the study illustrate that such behavior does not necessarily lead to a relative increase in the total level of the public good "environmental protection."[156] Moreover, under

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[156] For this, Hoel (1991) assumes that the other donors do not change their "selfish" payoff-maximizing behavior.

certain circumstances, unilateral actions may even lead to a comparatively low quantity of the public good.

Balland and Platteau (1997) explicitly take into account ecological thresholds that contribute to nonconvexities in the payoff functions of the donor countries. Against this background, noncooperative interaction between donors can lead to multiple equilibria with different levels of total contributions and thus conservation. When, for example, a low-level and high-level equilibrium are identified, the stakeholders have to coordinate to attain the high-level equilibrium. Aside from the coordination problem, the authors demonstrate that depending upon the impact of the threshold on the benefit function, there is a potential for incentives for substantial unilateral funding of biodiversity protection, i.e., incentives to free ride are less pronounced.

Regarding the context of privately provided public goods, pure public goods are often bundled together with impure public goods or private goods (Cornes and Sandler 1984). Analytical studies show that the gap between the level of protection attained in the noncooperative outcome and the optimal level decreases with an increasing proportion of the private good relative to the public good (Sandler and Hartley 2001). In the context of biodiversity protection, the concept of joint supply suggests that unilateral funding for conservation measures may generate both benefits that accrue only to the donor (private goods) and benefits that accrue to all countries (global public goods). The increasing importance of exclusive joint benefits (private goods) thus influences the incentive to spend financial resources on the provision of a pure public good.

In addition to these theoretical analyses, general *empirical descriptions* of multilateral and unilateral funding identify the practical strengths and weaknesses of both mechanisms: the advantages and disadvantages of *multilateral funding* are in essence related to its size. Since the institutions of multilateral funding manage a large stock of financial resources, they can control the allocation of transfers more effectively across a broad geographic scale. In this way, the imbalanced international assistance across the recipient regions resulting from decentralized unilateral transfers is avoided (Lapham and Livermore 2003). In addition, multilateral funding can help to exploit economies of scope in solving cross-border economic issues (Kanbur 2002).

Furthermore, Kölle (1995: 151f.) argues that a multilateral funding institution possesses bargaining power in bilateral negotiations with a resource country. As compared to a unilateral funding, the multilateral institution is able to negotiate a price for conservation activities that is close to marginal cost. When, in this regard, the multilateral institution acts on behalf of the donors, the money they provide for conservation is, ceteris paribus, used more cost effectively.

Nevertheless, donors may have different preferences. The need to aggregate these preferences and reconcile the donors' interests within the multilateral institution is a potential weakness of multilateral funding. Difficulties in reaching the consensus on the agenda also influence the funding of conservation projects. In this respect, multilateral funding is often considered bureaucratic and inflexible (Lapham and Livermore 2003).

In contrast, studies argue that *unilateral funding* is more flexible, less bureaucratic, and potentially more effective, since it is solely based upon the priorities of the two countries involved in the negotiations. Decision making is not inhibited by the need to reconcile differing agendas. Furthermore, donors often focus their unilateral assistance upon specific resource countries that are in geographical proximity and/or to which they have historical ties. An existing relationship with a specific resource country often represents the basis for an effective and continuing collaboration. In this case, a donor country may learn about the threats to biodiversity and socioeconomic needs in such a resource country more effectively and can therefore devise more accurate assistance (Lapham and Livermore 2003).

From the perspective of a donor country, the optimal choice between participation in a multilateral arrangement or the provision of funds on a unilateral basis cannot be generalized, but, rather, depends upon the nature of the spillovers/the international public good at stake. Overall, a complementary relationship between unilateral and multilateral funding apparently applies from the point of view of both the donors and resource countries as recipients.

Multilateral cooperation manifested in international agreements is frequently considered a global public good on its own. Kaul et al. (1999) distinguish (1) *final global public goods*, which are outcomes that directly generate benefits, from (2) *intermediate global public goods*, i.e., international regimes that contribute to the provision of the final global public good. In the context of international biodiversity policy, multilateral agreements represent intermediate goods that contribute to the provision of biodiversity as a final public good. Given that donors make the additional unilateral contributions, the provision of ecosystem services of a global importance is not dependent upon multilateral coordination as the intermediate public good.

4.1.2 International Protected Area Policy: Empirical Evidence

Given the theoretical background on international protected area policy, I study how official compensation or transfer payments manifest themselves in practice in multilateral transfer mechanisms and official/private unilateral transfers (Section 4.1.2.1). Subsequently, I describe what can be considered to be the current outcome of this policy (Section 4.1.2.2).

4.1.2.1 Multilateral Transfer Regimes and Unilateral Transfers

Protected areas are often perceived to be a cornerstone in the strategy for the effective conservation of biodiversity. Following the concepts described above, this section provides a review of international agreements that simultaneously arrange (1) protected area measures in (developing) countries hosting biodiversity of a global importance and (2) transfers that the donors grant for biodiversity conservation in excess of the domestically optimal level. First, I investigate *multilateral* arrangements, i.e., many countries donate and are usually represented by an international donor institution that enters into contracts with the individual resource countries for (additional) conservation measures. In the second step, I analyze transfers for conservation that individual donor countries offer on a *unilateral* basis.

Both types of arrangement refer to *official spending*, i.e., the donors belong to the public sector. In addition, private nongovernmental actors also donate money for conservation in the developing world. I briefly consider *private giving* at the end of the section.

International Agreements on Protected Areas and Transfer Mechanisms

There are a number of national regulations and international agreements that deal with the protection of biodiverse ecosystems directly or indirectly address the allocation of land areas among different uses (see Section 2.2). Together, these agreements form a developed and heterogenic system that makes use of different instruments and institutions.

Regarding the coordination among sovereign countries on an international level, a considerable number of *international environmental agreements* (IEAs) contain provisions on protected area measures. Most of these agreements originate from before the CBD and do not usually address the conservation of global biodiversity specifically, but, rather, aim at the protection of specific endangered species or specific forms of habitats on a regional or global level. Examples are the African Convention on the Conservation of Nature and Natural Resources, the Bern Convention, the Conservation of European Wildlife and Natural Habitat, the Bonn Convention on the Conservation of Migratory Species of Wild Animals, and the Antarctic Treaty (Matz 2003; Mulongoy and Chape 2002).[157]

Most of these agreements neither establish an individual mechanism to offer international transfers in return for the maintenance of global important eco-

[157] Furthermore, initiatives have been started, for example, in the United Nations framework. This is the UNESCO Man and Biosphere (MAB) Programme (Mulongoy and Chape 2002).

system services nor do they make use of existing mechanisms. Accordingly, most of the agreements do not represent compensation agreements of the type discussed above. Of those agreements that address the establishment and the management of protected areas, only three of them have implemented their own transfer mechanism or established a link to an existing mechanism (Matz 2003). These are

- the Ramsar Convention on Wetlands of International Importance (RC),
- the World Heritage Agreement(WHC), and
- the Convention on Biological Diversity (CBD).

The RC and the WHC both follow a listing approach, which means protected areas are specifically registered under these agreements as Ramsar Wetland Sites and World Heritage Sites (Chape et al. 2003). The CBD does not arrange for an agreement-specific network of protected areas and therefore does not pursue a listing approach. Moreover, it addresses the entire national system of protected areas in its signatory countries (CBD Art. 8a).

For both the RC and the WHC, donors have established treaty-specific environmental funds as transfer mechanisms. This is the *Ramsar Small Grants Fund* (SGF) and the *World Heritage Fund* (WHF). Both funds operate with a relatively small budget. Over the last decade, the SGF has, on average, transferred $0.3 million per annum (Ramsar 2002).[158] Data from the WHC (2002) shows that the WHF transferred $2.3 million in 2000 and $2.8 million in 2002.[159] However, the WHF not only offers international assistance for natural sites (which are rich in biodiversity) but also for cultural sites. Consequently, the amount of total WHF transfers is not a good estimate of the resources invested in biodiversity, since cultural sites include man-made and urban sites that are usually of little importance to biodiversity. Overall the SGF and the WHF regard themselves as having a catalytic role: they assist signatory countries in relatively small-scale projects in order to obtain funding for larger projects from other (international) donors (Matz 2003).[160]

The CBD's mechanism is the *Global Environment Facility* (GEF). As an international funding institution, the GEF is not confined to the issue of bio-

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158 In Ramsar (2003), the grant amounts are originally denoted in current Swiss francs. I converted them into current US dollars.

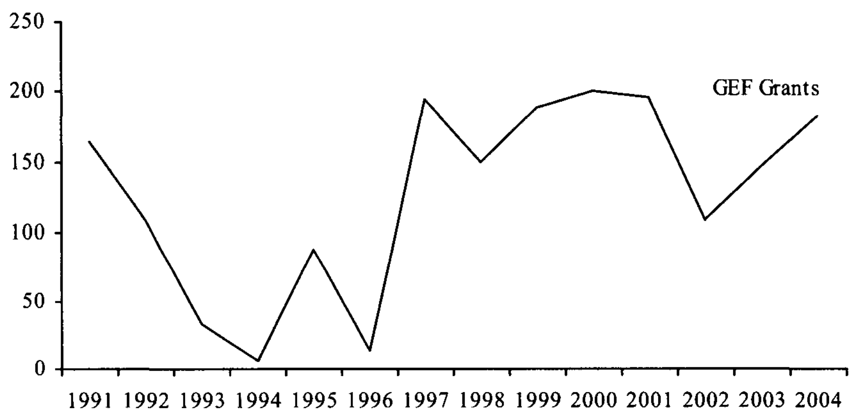
159 According to Spalding (2002), the total annual budget of the WHF is approximately $3.5 million.

160 The UNESCO Man and Biosphere Programme (MAB) also addresses protected areas according to a listing approach (World Network of Biosphere Reserves). Nevertheless, in contrast to the other regimes, the MAB network is not governed by an international agreement and does not have its own transfer mechanism (Matz 2003). For this reason, the program is not investigated any further here.

diversity but serves several international agreements that address various global environmental problems. It came into existence in 1991, i.e., before the CBD was signed. The aim of the GEF is to assist developing countries and transition countries in protecting the environment and promoting environmentally sound resource use and sustainable economic development. Sections 4.2 and 4.3 investigate the GEF policy on funding biodiversity projects of various types in more detail.

To describe the financial resources the GEF provides for biodiversity conservation, I use official data from the GEF online project database. In order to assess the annual total of GEF transfers for biodiversity, I focus upon the information indicated regarding the financing of the individual *biodiversity projects.* I calculate the total amounts by adding up the GEF project grants for the year of the project's approval. For simplicity, I regard these additions as the financial resources annually provided and illustrate them in Figure 17.[161]

Figure 17:
GEF Grants for Biodiversity Projects



Note: Total approved grants per annum in current millions of dollars. Figures represent GEF project grants calculated over the year of the projects' approval.
Source: GEF (2005), own calculations.

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161 By this, it is implied that payments are transferred up front when the project starts. In practice, however, the payments are actually disbursed over the entire project duration. Accordingly, the figures derived include some distortions. Nevertheless, since detailed information on actual expenditures in each fiscal year is not available, the figures derived represent a good approximation.

The figure indicates that after some volatility in the first years, when the GEF was established and the link between the GEF and the CBD was implemented, approximately $200 million were mobilized per annum. Furthermore, while the financial resources seem to have increased slightly but steadily in the late 1990s, this trend has not continued in recent years.

Not all of the GEF resources used for biodiversity conservation are explicitly directed to *protected area measures*, since other conservation instruments are also employed in the projects (see Section 4.2.2).[162] A study by the World Bank (2005) classifies the previous GEF transfers according to their use for protected area measures. It turns out that, on average, $131 million are used per annum for the establishment and management of protected areas. This result confirms the findings of a previous study on GEF biodiversity projects that were approved between 1991 and 1995 (World Bank 1995). Here, it turns out that 50 percent of the GEF grants are invested in protected areas. In contrast to these findings, other authors observe that the attention in international funded biodiversity projects has recently shifted away from protected area management towards integrated conservation and sustainable use projects (Perrings and Gadgil 2003; Lapham and Livermore 2003). Section 4.2.3 addresses these aspects in a detailed analysis of more than 600 GEF projects.

To conclude, of the multiple IEAs that address protected areas measures, only a few have established a link to mechanisms of transfer in order to create incentives for the developing countries to internalize the positive externalities from their globally important biodiversity endowments.

In the three regimes identified, transfers are resource flows from donor countries to resource countries, which both represent the signatory parties of the corresponding IEA with its associated mechanism. Donors usually offer transfers on a conditional basis, i.e., the money provided is earmarked for projects that address biodiversity conservation. Furthermore, the transfers are effectively intergovernmental grants and they are mainly given in cash. There is also some evidence of in-kind transfers, i.e., noncash transfers of technology and knowledge. With respect to the total size of the resources transferred, the three mechanisms of transfer differ substantially. The GEF, as the CBD's mechanism, plays the most important role in quantitative terms.

In addition to these mechanisms, further multilateral funding sources are *United Nations organizations*, such as the United Nations Development Programme (UNDP) and the United Nations Environment Programme (UNEP), and the

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[162] For example, GEF grants are invested in institutional capacity building in developing countries, which, however, may indirectly contribute to the enforcement of the strict protection of ecosystems. In other projects, GEF grants assist the implementation of environment-friendly management of natural areas, but extensive human uses are not explicitly excluded in this context.

World Bank. While these international institutions are not directly linked to IEAs, they are often engaged in the cofunding of GEF projects (see Section 4.2). For a description of the multilateral funding of biodiversity conservation offered outside the GEF framework, see Lapham and Livermore (2003) and World Bank (2003).

According to its own indications, the World Bank group mobilized a total of $4.3 billion for biodiversity conservation between 1988 and 2003. Three quarters of this have been made available for the management of protected areas. This corresponds to a total of $3.2 billion for the protected area portfolio (World Bank 2003). However, the figures include funding through the GEF; without this the total net contribution by the World Bank group in nominal terms is about $1.6 billion (World Bank 2003: Fig.1). I use these figures to roughly calculate the annual average transfer amount. The World Bank, on average, transfers approximately $143 million per annum for biodiversity in general and approximately $107 million for protected areas.

While the three multilateral mechanisms, the SGF, WHF, and GEF, make IEA-specific grant payments, the World Bank has transferred nearly $968 million out of a total of $1.6 billion on a loan basis through the International Bank of Reconstruction and Development (IBRD) or as credits through the International Development Association (IDA) (World Bank 2003). In contrast to the GEF grants, loan payments, which the resource countries have to repay (to a large extent), do not seem to be consistent with the concept of international transfers for the internalization and public good provision described in the previous section. Therefore the World Bank policy is not analyzed in detail in the following.

Finally, the three mechanisms described all have a global scope. In addition, transfer mechanisms are also established on a regional level, namely in the European Union. The *LIFE-Nature fund*, which is financed by EU member states, supports conservation projects in the context of the *Natura 2000 network.* From 2000 to 2004, this fund disbursed a total amount of €300 million (EC/EDG 2003).

Official Assistance through Unilateral Transfers

Aside from mobilizing financial resources within a multilateral framework, any country that is willing to contribute to the conservation of global biodiversity can conclude contracts for protection projects and make official transfers on a unilateral basis. Unilateral transfers are by definition decentralized; they are made available by both official donors and private nongovernmental donors. Furthermore, donors design them in several different forms, such as conditional grants, noncash transfers of technology or knowledge, debts-for-nature swaps, or loans with a grant element. For these reasons, it is difficult to completely describe the unilaterally provided funds in total.

To assess official funding by major developed countries, I analyze data given in the *OECD Creditor Reporting System* (CRS) (OECD 2005). This database contains information regarding official unilateral transfer flows from 22 developed countries that are listed in the OECD Development Assistance Committee (DAC), including flows from the European Development Fund (EDF). Developing countries and the countries in transition are the transfer recipients. The entries indicated in the database describe individual financial flows, which are predominately grants, such as Official Development Assistance (ODA) or Official Aid (OA); in some cases the flows also represent loan payments.

To describe the total amount of unilateral transfers per annum, I calculate the individual flows (presented in current dollars) over the years in which they have been committed.[163] As far as the flows represent loans, only their grant element is included in the figures. Depending upon how the financial flows that address biodiversity conservation are segregated from flows that address other purposes, the transfer amount ranges between less than $200 and approximately $900 million per annum.

The estimate for the lower amount of total transfers is based upon the OECD CRS classification of financial flows by funding purpose. One explicit funding category in this classification is "biodiversity." Figure 18 describes the amounts of annual transfers that donors have made available unilaterally for this purpose. The figure indicates that the unilateral funding has developed in a similar way to the path of the GEF funding: after some volatility in the early 1990s, the resources provided annually increased in the second half of the decade, but the margin of this increase has declined in recent years. The total transfer amount is presently approximately $200 million in current dollars.

The OECD classification of flows in the CRS database also includes several other funding purposes, such as "biosphere protection" or "site preservation," that may also contribute to biodiversity conservation. Consequently, the financial flows illustrated in the figure more than likely understate the actual unilateral transfers for conservation. In order to capture the cross-section impact of the conservation of biodiversity (and, more generally, environmental protection) upon projects of international assistance, the OECD introduced a scheme of indicators called the "*Rio Markers.*"

These indicators mark transfer payments that relate to activities addressing the global environmental problems that are dealt with in the Rio conventions. The flows that are marked in this way either address an individual environmental

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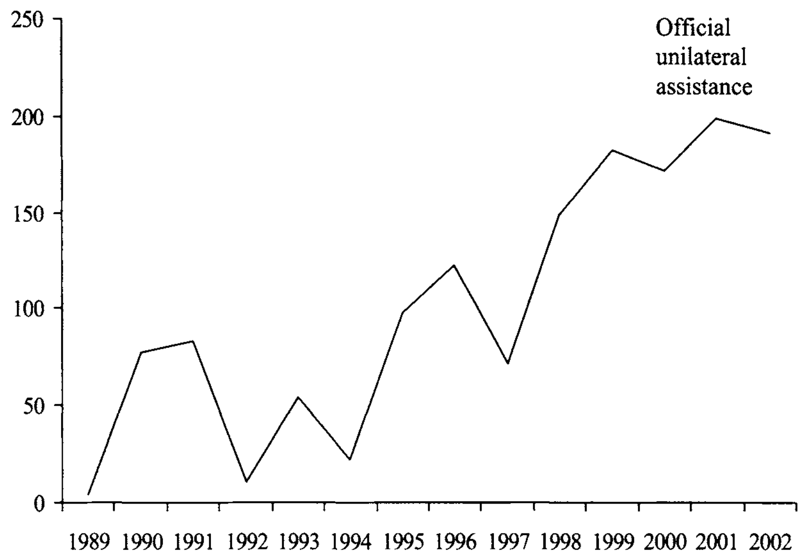
163 I again choose this approach, since it is not possible to identify the actual disbursement of the transfers over time

Figure 18:
Unilateral Grants by OECD Countries for Biodiversity Conservation



Note: Total approved grants per annum in current millions of dollars. Figures represent grants by OECD countries calculated over the year of projects' approval.
Source: OECD (2005), own calculations.

problem, such as biodiversity loss, climate change, or desertification, or several of them simultaneously in integrated projects (Lapham and Livermore 2003).

Table 11 indicates the annual total of flows that deal with "biodiversity." The right-hand column describes the annual total of flows that address biodiversity only. I again calculate the figures for individual flows over the years in which they were committed. The markers have only been assigned to flows since 1998. The flows for 2001 and 2002 seem (so far) to be incompletely recorded in the database. Consequently, figures are presented for three years only. From these figures, a trend in funding cannot be identified yet.

Since the amount for biodiversity funds in total is approximately twice as high as the total amount for flows focused upon biodiversity only, its conservation is obviously, to a large extent, dealt with in connection with measures in climate policies and policies to fight desertification. Accordingly, if this upper estimate of the total amounts were used to describe the unilateral transfers for protected area management, it would certainly overstate the amount actually offered for this purpose.

Table 11:
Official Transfers by OECD Donors Marked by the Rio "Biodiversity" Marker

Year	Total of flows Rio "biodiversity" marker	Total of flows Rio "biodiversity" marker only
1998	895.272	391.271
1999	783.659	466.189
2000	781.017	363.233

Note: All figures in current millions of dollars. Total of flows calculated over the year of commitment.

Source: OECD (2005), own calculations.

To conclude, although it is difficult to identify the official unilateral transfers that address biodiversity conservation, the data in the CRS database imply that OECD countries currently provide between $200 and 900 million per annum for activities in transition and developing countries.[164] Again, this figure more than likely overstates the funds that are actually made available for protected area management in the resource countries, since it includes payments for a variety of other activities in conservation and in environmental protection.

International Private Giving in Developed Countries

So far, this study has perceived payments (and noncash transfers) by developed countries to be transfers on an intergovernmental level. In addition to the official assistance, the private sector provides resources for biodiversity preservation in the developing world on a voluntary basis. This is supported by anecdotal evidence on *nongovernmental organizations* (NGOs), *foundations* that conduct international conservation activities, and *multinational firms* that implement nonprofit programs for conservation. Furthermore, private donors have sometimes used debt-for-nature arrangements to finance conservation in developing countries.

The data on funds transferred by private donors is sparse and not systematically recorded. Accordingly, it is difficult to reliably assess how much the private sector offers to the developing countries. Results from a study on the international giving by philanthropic foundations can offer an approximation (OECD 2003c). The study investigates the annual private giving in major developed

164 These figures do not include donors outside the OECD. However, little is known about how many transfers oil exporting countries or newly industrialized countries, for instance, spend on environmental protection outside their own borders. Overall, it seems reasonable to assume that official assistance by OECD countries represents a very large portion of the actual transfer payments.

countries for the purpose of development cooperation. The total annual amount of international private giving between 1994 and 2000 is assessed at \$1.0 to \$3.1 billion. Most of these financial resources originated from US-based foundations and most of the resources are used for purposes other than conservation.

Considering the private donors by regions, figures for *US-based foundations* in 2000 indicate that conservation activities, including activities involving natural resources and wildlife, received 6.7 percent of the total international giving. This corresponds to funds for conservation activities of approximately \$163 million. The OECD study does not document contributions by *European foundations* in the same quality. Based upon estimated figures, these foundations provide \$350 million for activities outside Europe. When roughly applying the same share of international giving for conservation activities identified for foundations in the United States to foundations in Europe, the estimate for private giving in Europe for international conservation is \$23 million per annum.[165]

To resume, even though the data on private transfers is poor in comparison to the data on official transfers, empirical evidence suggests that the size of private giving is below that of official assistance. When calculating the annual flows from the three described sources, the *total annual amount of international transfers* for biodiversity conservation ranges from roughly *\$0.6 to about \$1.0 billion.*

These figures can be compared to estimates of the total expenditures for protected areas worldwide. For example, James et al. (1999b) estimate that these expenditures amount to \$6.0 billion per annum.[166] However, when regarding the regional allocation of the funds, it turns out that 88.4 percent are expended in the developed regions. Consequently, only \$0.695 million (in 1996 dollars) are invested in the protected areas located in the biodiversity-abundant developing countries. While this figure seems to coincide with the range of the size of the international transfer flow derived above, the fact that the figure for the developing countries includes domestic investments in their own protected areas has to be taken into account. Furthermore, as discussed above the range derived refers to transfers that do not exclusively target protected area management, but rather biodiversity conservation in general.

4.1.2.2 Policy Objectives and Outcomes: A Global Network of Protected Areas

Given the description of the current international policies on biodiversity and protected areas in its legal and financial dimension, I study how the outcome of

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[165] Figures for international giving for biodiversity conservation by Asian foundations can hardly be identified (OECD 2003c).

[166] The figure is based upon 1996 dollars and considers a global network of protected areas covering 1.3 billion hectares (James et al. 1999b).

such policies manifests itself in a network of protected areas. First, I summarize data on the actual extent of protected area systems worldwide. Based upon this, I review recent studies on the size and estimated costs of the effective global network the policy is aimed at.

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 al. 1994).The World Conservation Union (IUCN) and the Conservation Monitoring Centre (WCMC) have compiled data on protected areas in the *United Nations List of Protected Areas* (Chape et al. 2003).

When assessing the extent and scope of the global network of protected areas, problems arise due to a lack of reporting by individual resource countries and differences in the countries' definitions of protected areas. For this purpose, the IUCN has developed a protected area classification that defines seven categories of protection that vary in the degree of exclusion from human use. Protected areas of category (Ia) and (Ib) display the strictest protection from human interference. In all subsequent categories, the exclusion is relaxed stepwise. At the end of the spectrum, category (VI) allows for some sustainable resource extraction and ecosystem modification. Although the *IUCN classification* is widely used meanwhile, there are still designated protected areas that are not classified according to this system (IUCN 1994; Chape et al. 2003).

The United Nations list makes use of the ICUN categories and aggregates national area protection systems into 15 regions that are defined by the IUCN World Commission on Protected Areas (WCPA). Table 12 presents summarized figures. It is shown that approximately 1.9 billion hectares worldwide are put under some form of protection. This corresponds to 11.5 percent of the global land surface (Chape et al. 2003; Mulongy and Chape 2003).

The *individual regions* show differences with respect to both the extent of protected areas (relative to the regional land surface) and the extent of their protection. The figures in the first column of Table 12 describe the absolute extent in million hectares. It illustrates that the level of protection is highest in the large regions that are rich in biodiversity, such as North America (including Mexico) and South America. The regions differ considerably with respect to protection relative to the total land surface. The shares range from less than 5 percent to over 27 percent. Generally, it is difficult to identify a pattern that explains the distribution of shares. Roughly speaking, large shares can be observed for the regions with large biodiversity abundance and with substantial economic wealth. Examples are North America or Australia. Nevertheless, the regions sometimes consist of countries that show distinct endowments of natural resources and

Table 12:
The Current Global System of Protected Areas

Region	Protected areas (in total)		Protected areas (Cat. I–III)	Protected areas (Cat. IV–VI)
	In million hectares	In % of land	In % of protected areas	In % of protected areas
North America (incl. Mexico)	455.3	20.8	50.2	48.5
Caribbean	6.9	29.6	40.0	57.4
Central America	14.5	27.9	36.1	29.3
South America	396.3	22.2	22.1	25.6
North Africa and Middle East	127.3	9.9	18.3	77.0
Western and Central Africa	112.6	8.8	35.2	40.1
Eastern and Southern Africa	196.7	17.2	25.9	42.2
Europe	75.0	14.6	23.4	61.0
North Eurasia	181.7	8.2	28.2	53.6
East Asia	103.2	8.8	65.5	31.1
Southeast Asia	76.0	16.4	33.6	45.2
South Asia	30.9	6.9	23.2	58.8
Australia and New Zealand	118.7	14.8	49.6	50.3
Pacific	2.0	3.7	32.6	58.2
Antarctic	7.0	0.5	97.1	0.7

Source: 2003 UN list of protected areas (Chape et al. 2003), own calculations.

biodiversity, as well as different levels of economic wealth. Therefore, the hypothesis that there is a relationship between the structure of protected area systems, endowments, and economic wealth requires further investigation.

In order to study the *structure of protection* in the individual regions, I summarize the figures on the protected areas of the first four ICUN categories of stricter protection and determine their relative share (third column). I do the same for the protected areas of categories IV to VI, which show a lower degree of protection. I omit the share of unclassified areas (with a potentially even lower degree of protection).

For North and Central America, as well as Australia, a large extent of protection is combined with a large proportion of strict protection. In contrast, for South America and Southern Africa, the protected land surface is of a comparatively large size but the degree of protection is relatively low. The reverse is true of East Asia and the Antarctic, where protection is of a relatively small extent but of a comparatively strict form. For the Pacific region, South Asia, and the North Eurasia region, protection of both a comparatively low extent and low degree is observed.

In total, the aggregate figures on the quantities and quality of protected area systems provide little evidence on whether the countries adequately address the need for protection and whether these systems guarantee the long-term preservation of biodiversity when facing the multiple anthropogenic causes of biodiversity decline (SCBD 2004).

Financial Needs for Managing a Global System of Protected Areas

Given the figures on the international transfers for protected areas and the figures on the current extent of the global network of these areas, the question is whether the extent of protection indeed meets internationally agreed upon conservation targets and whether the transfers provided reach a level sufficient to assist effective conservation in the developing countries.

As mentioned earlier, the determination of conservation targets on a local level is quite challenging. Accordingly, it is even harder to aggregate local protection needs in order to determine targets on a global level. However, in order to provide practical guidelines, the *10 percent target* was advocated in the international biodiversity policies of the 1980s and 1990s. This target goes back to the Bali Action Plan that was released by the IUCN in 1982 and that, inter alia, recommended greatly expanding protected area systems (Miller 1994; Sanjayan and Soulé 1997). This paper influenced the 1987 *Brundtland Report*, which recommended that, for the effective preservation of biodiversity, the size of the global protected area network should be tripled. Since, at that time, approximately 4 percent of the global land surface had been placed under some form of protection, the recommendation was loosely interpreted to mean that 10 to 12 percent of the global land surface should be placed under some form of protection, although this was not defined any further (Soulé and Sanjayan 1998). In 1992, the Fourth Congress on National Parks and Protected Areas affirmed the 10 percent target (SCBD 2004: 45).

Some studies, for instance, Soulé and Sanjayan (1998), have critically reviewed this quantitative target. As a reaction, the qualitative approach of *representative protected area systems* on a national and regional level, the combination of which forms a global network, has been put on the political agenda. This objective was supported at the Fifth Congress on National Parks and Protected Areas in 2003 and at the 2004 meeting of the Conference of Parties to the CBD (COP7). In the decisions of the meeting, it was supposed that an effective network would directly contribute to the achievement of the *2010 target* (see Section 2.2).[167]

[167] In order to verify whether designated protected areas are representative of a specific region and whether global biodiversity is well captured in the global network,

Regarding the assessment of *financial needs* for the effective management of the global network, several studies derive (gross) costs of protection on a global level. These studies usually define the protection target as a percentage of the land surface or assume ad hoc targets in this regard.

In the following, I review four studies that have derived figures on the total (gross) costs of protection. I discuss to what extent the *cost estimates* can serve as approximations for the financial needs of the network of global protected areas. Since the literature typically assumes that developed countries finance their own protected areas by themselves, my focus falls upon financial needs for protected areas in the developing countries, including countries in transition:

- *James et al. (1999b)* argue that in each of the ten different continental regions, 10 percent of the land area (or a total of 1.6 billion hectares) should be strictly protected. In order to implement this target effectively, 15 percent of the land surface has to be placed under protection. Based upon a spreadsheet simulation, the authors calculate that protection to this extent is associated with total annual costs of $27.6 billion. On a regional level, $14.9 billion accrue to the six regions that constitute the developing countries and countries in transition. These figures contain (1) the costs of compensating local communities as landowners for their forgone revenues, (2) the costs of optional land purchases, and (3) the costs of managing the existing and newly established protected areas.[168]
- *Lewandrowski et al. (1999)* analyze the costs of setting aside land from agricultural use. In order to calculate the total economic costs of such a policy, they use a global, but regionally disaggregated, computable general equilibrium (CGE) model. The model describes eight economically defined regions. To consider the different land productivities within each region, the authors employ specific land use data. Finally, they define scenarios of 5, 10, and 15 percent reductions in the productive land endowments. The reductions are enforced in each region and for each different class of land productivity. The resulting annual protection costs on a global level are $45.5, 93.3, and 143.8 billion (all figures in 1990 dollars). The total costs for the three regions that represent the developing and transition countries are $16.1, 33.1, and 51.1 billion across the different scenarios.
- *Myers et al. (2000)* call for the protection of selected biodiversity hotspots. Their study provides some figures on protection costs, i.e., the financial resources needed for the safeguarding of the identified hotspots. Based upon

scientists involved in conservation planning use extensive local data and conduct global gap analyses (SCBD 2004).

168 The figures derived are presented in 1996 dollars. The latter types of costs are derived by extrapolation from observed land values and management costs.

ecological data, 25 hotspots are identified, which together cover 210 million hectares (or 1.4 percent of the global land surface). For the protection of all hotspots, a total of $0.5 billion per annum is needed. This figure is based upon the authors' (ad hoc) assumption that the protection of a single hotspot on average requires an annual amount of $20 million. Since 20 Hotspots are located in developing countries, the costs these regions incur are $0.4 billion.

- *Balmford (2003)* updates the figures presented in James et al. (1999b) but uses alternative estimates for land purchases. The author estimates the costs of managing terrestrial protected areas at an annual amount of $24.5 billion (in 2,000 dollars). When applying the share of the costs accrued to the developing regions used in James et al. (1999b), the extrapolated costs in these regions amount to $13.2 billion.[169]

When taking the cost figures as estimates for actual financial needs, I find that each of these studies has both its strengths and weaknesses. Regarding first the definition of protection targets, all studies except the study by Myers et al. (2000) essentially rely upon the assumption that one tenth of the land area in certain spatial classifications should be placed under some form of (strict) protection. Against the background of ecological and economic diversity among regions, the studies provide no justification for this assumption on economic and/or ecological grounds (Soulé and Sanjayan 1998).

In contrast, Myers et al. (2000) give detailed advice on where natural areas with an exceptionally high level of species diversity are located. The spatial expansion of the individual hotspots is, however, defined by purely biological criteria,[170] i.e., no economic considerations enter into this definition. From an economic point of view, the optimal size of a specific hotspot could be smaller or even larger, depending upon the benefits and costs of protecting a marginal unit of land.

Furthermore, Myers et al. (2000) assume equal costs of protection across the hotspots and thereby abstract from the fact that the selected hotspots expand across natural areas that vary significantly in size (0.2 to 35.6 million hectares). In addition, they are located in various countries that differ significantly in land values and management costs. These differences are not accounted for in the cost assessment.

Except for the study by Lewandrowski et al. (1999), the figures derived can be regarded as gross protection costs, since no private goods, such as tourism

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[169] Bruner (2003) also reviews and confirms that the costs of expanding and managing protected areas in developing countries amount to approximately $12 to $13 billion per annum.

[170] A biome is a hotspot if it hosts 0.5 percent of global plant species diversity on its area (Myers et al. 2000).

services that may be produced within protected areas and thereby generate some income from protection are taken into account.[171] The CGE approach applied by Lewandrowski et al. (1999) is a powerful tool for analyzing the economic adjustments that take place within an economy when the use of productive land is exogenously restricted. For this reason, the cost figures, in contrast to the figures in the other studies, also include secondary economic impacts due to changes in the relative prices that arise when protected areas are established. Otherwise, services generated for other sectors, such as tourism, are included in the simulation results.

A caveat to this study is, however, that it is not clear whether the natural areas that have already been designated as reserves are included in the scenarios or whether the model assumes zero protection in the benchmark. The study is not explicit on this point. In the latter case, the underlying protection objectives do not refer to the total land surface, but rather to the current agricultural area, which would imply that protection targets are much stricter than the numerical percentage indicates.

Finally, Table 13 summarizes the results of the studies. Generally, the figures derived represent rough estimates and should be treated with caution. However, it is remarkable that the figures consistently indicate that the *total costs of protection* rise as protection targets become more ambitious. By interpreting this as evidence of the figures' reliability, I conclude that the actual annual demand for funds to finance the worldwide protection of natural areas lies within the range of \$0.5 billion to approximately \$150 billion. When focusing upon the *costs of protection in the developing world*, I identify a range from \$0.4 to \$51.1 billion.

When using the cost estimates to determine the actual financial needs that developed countries have to mobilize as transfers, two aspects have to be considered. First, the cost estimates may have to be adjusted for the net costs by subtracting revenues from marketable goods (tourism services, genetic resources) generated in protected areas. Second, the spatial expansion and/or effective management of protected areas in the developing countries also generate additional benefits for these countries. Accordingly, they may also participate in financing the costs of (incremental) protected areas. Section 4.3 analyzes these aspects in more detail.

[171] One may argue that within strictly protected areas, no private goods can be produced since nearly every human use is excluded. However, note that according to 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.

Table 13:
Annual Costs of Protected Area Systems

Study	Protection target	Annually financial resources needed in billion $		
		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)	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	24.5	13.2	Extrapolated values, see James et al.

Both aspects relate to the participation of developing countries in cost sharing and effectively imply that the derived cost figures represent an upper estimate for the size of the transfer flow the international donor community ideally offers. A *comparison* of these figures with the above estimates of *official multilateral and unilateral assistance* from developed countries generally has taken into account the difficulties described in approaching the actual demand for financial resources. Nevertheless, the comparison suggests that the total resources that have been provided annually by developed countries fall substantially short of the required amount (James et al. 1999b; Bruner 2003).

On the one hand, this shortfall of financial resources is due to *policy failures within developing countries*. For example, the administrative entities of protected areas in these countries are often not properly endowed to carry out the effective monitoring and control of existing reserve sites; on the other hand, *incentive problems in the international cooperation* can be made responsible for the gap in the financial resources that donors make available for protection measures in developing countries. The analysis in Section 4.4 focuses upon this latter aspect.

4.2 Transfers by the Global Environment Facility (GEF)

Building on the theoretical findings on international spillovers from biodiversity conservation and international transfers (see Section 4.1.1), the empirical findings of multilateral and unilateral assistance presented in Section 4.1.2 indicate that the GEF plays an important role in the international policy on biodiversity and protected areas. Although the funding it provides has previously been relatively modest, the GEF receives political attention, since it serves as the financial mechanism of the IEAs, which address major global environmental problems (Fairman 1996).

Against this background, I first summarize empirical facts on the GEF as an international financing institution (Section 4.2.1). Based upon this, I study how GEF funds for biodiversity conservation are allocated and, in particular, how many funds are directed to protected area policies. For this purpose, I describe GEF expenditure on projects for biodiversity conservation in Section 4.2.2. Since my conceptual focus falls upon ecosystem protection and transfer payments, my further analysis concentrates on projects that support protected area measures (Section 4.2.3). Finally, I briefly discuss aspects of the sustainable funding of protected areas in the context of the GEF funding (Section 4.2.4).

4.2.1 The GEF as a Multilateral Financing Institution

From an economic perspective, the GEF is a multilateral institution that aims at the internalization of positive spillovers that (1) are generated from environmental resources in the developing world and (2) are of global concern. GEF grants assist developing countries in their resource management, which generates positive spillovers to the developed countries (and the other developing countries).

Considering the external benefits of these environmental resources, GEF funds have previously been allocated to projects in four *focal areas*:

- biodiversity,
- climate change,
- international waters,
- ozone depletion.

In 2002, the scope of the GEF was broadened by the introduction of two new focal areas (GEF 2005b):

- land degradation,
- persistent organic pollutants (POPs).

The *global dimension* of biodiversity preservation is reflected by nonuse values (existence or bequest values) and option values. These values are generated from the preservation of living organisms that represent biological diversity on a genetic, species, and ecosystem level. Option values relate to the present and future use of the world's gene pool for biotechnological R&D (Perrings and Gadgil 2003).

Preservation also contributes to the maintenance of supportive ecosystem services and thereby assists the generation of indirect use values. The recipients of these values are primarily located at the local level. However, since these services are often essential to the economic capacities of developing countries and the living conditions of its people, the preservation of these ecosystem services is also indirectly of international concern. As shown by the provision of development assistance, the developed countries have an interest in the sustainable development of the poorer countries (Kanbur 2002; Jayaraman and Kanbur 1999). Thus, international assistance supports activities that preserve local ecosystem productivity, even though the benefits generated therefrom are primarily appropriated by the local population.

Features of the GEF

The GEF as an international institution displays specific characteristics in a legal, institutional, economic, and political dimension that distinguish it from other institutions that regulate international relations (Silard 1995; Boisson de Chazournes 2003).

In essence, the GEF is a trust fund with its own body of governance and a network of arrangements with other institutions. In 1989, an initiative to establish an international funding mechanism that supports developing countries in their actions on global environmental problems led to the GEF's creation as a multilateral mechanism of transfer. The *three-year pilot phase* of the GEF started in 1991. Initially, conflicts between the stakeholders involved hampered the implementation and operation of the GEF. These stakeholders were (1) the governments of developed and developing countries, (2) the multilateral institutions of the *World Bank*, the *United Nations Development Programme* (UNDP), and the *United Nations Environment Programme* (UNEP), and, finally, (3) NGOs in politics and science that showed an interest in international environmental assistance (Fairman 1997). Due to the conflicts, the governance rules of the GEF were reconstructed in 1994 (Sjöberg 1999).

The governing bodies of the GEF are the *Participants' Assembly*, which includes the governments participating in the GEF as donors and recipients. The task of the assembly, which meets every 3 to 4 years, is the specification and adaptation of a general strategy, including decisions on the continuous income of the fund.

The main executive organ is the *Council*. It meets twice a year and decides upon operational policies and programs, including the approvals of GEF projects (Boisson de Chazournes 2003). The Council is composed of representatives from

16 developing countries and two transition countries as recipients and 14 developed or donor countries. There is a call for decisions in the Council to be taken by consensus. If a consensus is not attainable, proposed regulations must have the support of at least 60 percent of all participants, as well as of the countries that provide 60 percent of the GEF's funds. In this respect, a double majority system guides the decision making in the Council. A *Chief Executive Officer* heads the council (Fairman 1996; Sjöberg 1999; GEF 2004b).

Further institutions are the *GEF secretariat*, which supports the operations, and the *Science and Technical Advisory Panel (STAP)*, which reports to the Council. More especially, the STAP gives advice on overall operational strategies and programs and provides expertise for selected project proposals. Finally, the *Implementing Agencies*, i.e., the World Bank, the UNDP, or the UNEP, assist a country that endorses protection activities on its territory and applies for funding in the application and implementation process. In this case, the Implementing Agency is accountable to the Council for activities funded by the GEF (Boisson de Chazournes 2003; Fairman 1996).

The Allocation of Funds

Since the pilot phase, the World Bank has held GEF resources in trust. For its pilot program, international donors provided the GEF with about $1 billion (Silard 1995). Since GEF disbursements are made on a grant basis, the GEF trust fund has to be replenished from time to time. Over the last decade, there were three replenishments, each of which raised $2 to 3 billion for the subsequent four-year period. I consider the coordination and decision making regarding the financing of the GEF in more detail in Section 4.4. Donors to the GEF have scheduled negotiations for the fourth replenishment (GEF-4) for 2006.

With regard to the GEF's *biodiversity focal area*, potential recipient countries that host valuable biodiversity endowments are eligible for project funding by the GEF if they are parties to the CBD. A bilateral agreement between the governing bodies of the GEF and the CBD has formally established the link between the two institutions (UNEP/CBD 1996a).

When the GEF commenced operations, the stakeholders in the decision-making process intended to allocate the funds of the pilot program across the original focal areas in accordance with a general rule of thumb that stated that 30 to 40 percent of total funds should be allocated to biodiversity projects (Pearce 1995: 143). Due to the recent enlargement of the GEF focal areas, current projections imply that, in the future, only about 30 to 32 percent of total funds will be allocated to biodiversity projects (GEF 2002a). In the late 1990s, the GEF provided total grants for biodiversity projects of approximately $200 million per annum (see Section 4.1.2.1). According to the projections, the GEF grant amount for biodiversity will increase to $250 to 290 million per annum at the end of the GEF-3 period (GEF 2002b).[172]

4.2.2 The GEF Policy on Biodiversity

Given the aggregate figures concerning grants on biodiversity conservation, I study how these GEF funds are allocated on a project level. In the GEF framework, the assistance by the industrialized countries takes place in the form of financial payments. Transfers by the GEF are typically based upon a (medium-term) contract between the resource country, represented mainly by public sector authorities, and the international donor community, represented by one of three implementing agencies, i.e., the World Bank, UNDP, or UNEP. The subject of the contract is conservation activities within the resource country that are financed by both domestic and international sources. The stakeholders bundle together both conservation activities and the associated flows of financial resources in form of a project.

For the empirical analysis of these projects, I use data on project funding that is presented in the *GEF Project Base.*[173] More specifically, I consider database entries for *652 biodiversity projects* that have been approved during the period of 1991 to 2005. When calculating the costs indicated for each project over all of the projects, this amounts to *$5.35 billion.*[174] The grants provided by the GEF amount to a total of *$1.78 billion*. Domestic sources and international donors other than the GEF finance the remaining *$3.57 billion*. Regarding the latter figure, the GEF database does provide any information as to which countries or institutions participate in the cofinancing and how many resources each cofinancier makes available in the individual projects.

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172 Several official and nongovernmental studies have evaluated the operations of the GEF and its financing of environmental protection. The GEF itself regularly commissions an Overall Performance Study (UNEP/CBD 2002). These evaluations have addressed multiple issues. Criticism has, inter alia, concerned the lack of a definition of an overall protection strategy and the selection of projects that have resulted in this regard. Furthermore, local communities that have been affected by project actions have participated insufficiently in the planning and implementation of the projects, which is said to finally limit the effectiveness of the project investment. As a consequence, there is evidence of increasing private sector involvement in the projects of the recent GEF periods. In this context, there are also calls to make the GEF project cycle procedures less complex; i.e., to streamline procedures and make them more transparent.

173 The analysis presented is based upon data from the database hosted at http://www.gefonline.org/ home.cfm). Downloads were made between January 2004 and April 2005.

174 Similar to the related literature, the calculation considers costs in current dollars.

In addition to the GEF database, the World Bank, which frequently represents the Implementing Agency in a project process, hosts its own database on GEF projects, the World Bank GEF Project Database, which is, however, confined to GEF biodiversity projects assisted by the World Bank. The database also lacks detailed information regarding donors in the cofunding of the projects; nevertheless, it contains additional information on cofinancing by the *IBRD* and the *IDA*: only 36 projects of all GEF biodiversity projects show cofinancing by either the IBRD or the IDA. Grants and loans by the World Bank institutions together amount to *$612.43 million* as part of the cofunding in the identified projects; *the IDA* provides *$366.07 million*[175] of this. While this amount seems quite small compared to the GEF grants, it needs to be highlighted that the World Bank finances plenty of projects on biodiversity conservation outside the GEF framework (World Bank 2003; see also Section 4.1.2.1).

The GEF Operational Strategy and Its Expenditures on Biodiversity

A specified operational strategy guides the funding of projects by the GEF. The elements of this strategy are (1) operational programs, (2) short-term measures, and (3) enabling activities. For the biodiversity focal area, four *operational programs* (OPs) are defined. Each of them addresses a specific type of ecosystem (GEF 2005c):

- arid and semi-arid zones (OP 1),
- coastal, marine, and freshwater resources (OP 2),
- forests (OP 3),
- mountains (OP 4).

Recently, conservation and sustainable use of biological diversity important to agriculture has been introduced as a new operational program (GEF 2005c).

The projects funded under these operational programs represent the core of the GEF's biodiversity portfolio. *Short-term measures* add a certain degree of flexibility to the strategy, as they support activities that are not originally part of the operational programs but that are nevertheless favorable, since they are cost-effective and address urgent conservation needs or promising opportunities in this respect.[176]

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[175] This figure does not include a World Bank credit of $93.9 million provided to the 2001 Chinese "Sustainable Forestry Development" project that was invested in the "natural forestry management/plantation implementation" component of this project. If this component were included, the total project cost would no longer coincide with the cost figures provided in the GEF database.

[176] The protection and rehabilitation activities after the Galapagos oil spill in January 2001 are an example of such short-term measures.

Enabling Activities aim at the assistance of developing countries in establishing national strategies and plans to preserve their biodiversity endowments. These activities include policy analyses and the development of conservation strategies and actions plans. Furthermore, enabling activities support the inventory, compiling, and disseminating of the information needed for communication on a national and international level. Capacity building is a potential follow-up to enabling activities and is covered by the operational programs (GEF 2000, 2005c).

The projects, which are funded as part of one of the three strategy components, belong to one of three specific *project types*: (1) regular or full-size projects, (2) medium-size projects, and (3) enabling activities.[177] The type of the project is generally correlated to its financial size: the GEF's grant for medium-size projects must not exceed $1 million. For enabling activities, the maximum size of the GEF grant is $0.45 million. Furthermore, the procedures for planning, designing, and approving the project vary among the project types.

Figure 19 illustrates the density distribution of the share of the GEF's grant relative to the total cost of the project. Given the different intervals for the GEF share, the number of projects in each of them is differentiated according to the project type. The right-hand column in the figure indicates that for more than 200 of the 652 projects, the GEF covers the complete project cost (100 percent). However, these completely financed projects are primarily small enabling activity projects. If projects of this type are neglected for a moment, the figure further implies that funding by the GEF for the remaining medium- and full-size projects is, on average, approximately 50 percent, with a substantial variance between the projects. Regarding full-size projects, it turns out that their cost can amount to about $100 million.

In addition, I calculate figures on *cost coverage* and shares in the project financing. The share of the aggregate GEF grants relative to the aggregate project costs is 33 percent. This percentage on the aggregate level is compared with the mean share of the GEF funding in the projects. The average cost coverage by the GEF (68 percent) is more than twice the GEF's share of aggregate cost coverage. This implies that the GEF covers large parts of the project costs in many small projects. Large projects display a distinct financing pattern; they rely upon substantial funding from sources other than the GEF.

GEF Grants for Biodiversity Projects: An Empirical Description

Given the information on the structure of project financing, I study how projects and GEF grants are allocated across recipient countries. The issue of financing is

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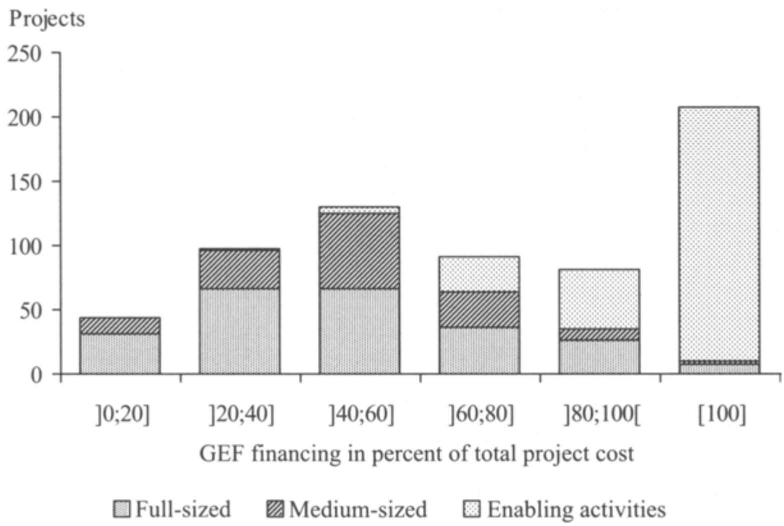
177 Consequently, enabling activities constitute both a project type and a component of the GEF strategy.

Figure 19:
Coverage of Project Cost by the GEF According to Project Type



Source: GEF project database, own calculations.

examined in Section 4.2.3. I first investigate the distribution of projects and the allocation of GEF project grants according to criteria from the realm of the natural sciences. In addition, I provide a description using socioeconomic criteria.

Considering the geographical and ecological criteria, Table 14 shows how projects and financial resources are allocated across the five *continental regions* as they are classified in the GEF database.[178] It turns out that Africa hosts the largest portion of GEF projects (217). For both Asia and Latin America, an identical number of projects (166) are observed. Comparatively fewer projects have been carried out in the transformation economies in Europe and Continental Asia (79).

I again calculate the figures on project costs and GEF grants. This indicates that more than $1.77 billion is invested in activities carried out in Africa. A comparatively smaller investment amount is indicated for Latin America ($1.57 billion) and Asia ($1.27 billion). Activities in the transformation economies attract $0.42 billion.

178 This classification seemingly relies more upon political than ecological criteria (for instance, the classification of biomes).

Table 14:
GEF Biodiversity Projects and their Regional Allocation, in Millions of Current Dollars

Region	Latin America, Caribbean	Africa	Asia	Europe, Continental Asia	Global, regional
Number of projects	166	217	166	79	24
Aggregate of total project cost	1,566.7	1,765.7	1,274.3	417.1	328.0
GEF grants	544.3	511.2	426.4	160.0	141.8
Cofinancing	1,022.4	1,254.4	848.0	257.1	186.2
IDA financing	10.0	245.9	97.5	12.7	0.0
IBRD financing	98.5	0.0	121.9	1.0	25.0

Note: The projects were approved during the period 1991–2003. The monetary figures displayed are additions across projects.

Source: GEF project database, World Bank database, own calculations.

The GEF covers approximately one third of the aggregate project cost in Latin America and Asia. A comparatively smaller GEF share is observed for Africa, which implies relatively high level of assistance from other donors, since domestic funding in biodiversity conservation is relatively small in these regions (James et al. 1999b). The IDA in particular plays an important role in Africa; it covers nearly one fifth of the aggregate project costs. As far as projects in Asia are concerned, the World Bank finances roughly one fifth of the project costs. Funding is, however, primarily offered through the IBRD. Latin America attracts comparatively little funding from the World Bank in both absolute and relative terms. Given the size of the GEF grants compared to the aggregate project costs, a lot of funding for conservation in this region seems to be raised from domestic and/or other international sources. Finally, Europe and Continental Asia display the highest relative GEF funding (more than 40 percent), although this funding, in absolute numbers, is the smallest among the regions. The World Bank provides funding for less than 5 percent of project activities in this region.[179]

While the figures in the table refer to the entire community of recipient countries, studies have identified certain countries that host an exceptional abundance of biodiversity: the 12 *Megadiverse Countries* (MDCs). Based upon this biogeographic classification, the political group of the 17 Like-Minded Megadiverse Countries (LMMCs) was formed (see Section 2.1.2). To examine the GEF policy on these overlapping country groups, I classify the community of GEF recipient countries in MDCs and Non-MDCs and describe the distribution of projects and the allocation of GEF grants according to these two groups.

[179] For completeness, the last column to the right covers projects that are not assigned to one specific region, but, rather, are cross-regional or global activities.

It turns out that 96 biodiversity projects (15 percent of the total of 652) are carried out in the 11 MDCs, which simultaneously represent recipient countries. The aggregate costs of projects in these countries amounts to $1.5 billion (28 percent of the total project cost aggregated across all 652 projects). The sum of GEF grants for projects in MDCs amounts to $0.46 billion, i.e., 26 percent of the total amount of grants. Regarding the group of the 17 LMMCs, a similar percentage is found.[180]

In order to assess whether MDCs or LMMCs attract more than proportional GEF funding, I first define the community of recipient countries as a benchmark. This group of countries consists of the 178 signatories to the CBD, excluding the 23 members of the OECD Development Assistance Committee. Consequently, MDCs represent 9 percent of the recipient community. The share of the LMMCs is 11 percent accordingly. I compare these percentages with those concerning the allocation of GEF funds. It seems as if the special role of MDCs is recognized in the GEF portfolio: a more than proportional share of grants is allocated to the MDCs. Nevertheless, the deviation from proportionality is not strongly pronounced. Consequently, without largely prioritizing the MDCs, the GEF biodiversity portfolio seems to be quite wide in order to support the conservation of representative biodiversity worldwide.

Evidence of representativeness can, inter alia, be derived from the ecosystem criteria that serve to define the operational programs (OPs) in the GEF biodiversity focal area. As mentioned earlier, the GEF defines four different OPs that refer to biological and geographic *ecosystem characteristics* of the site addressed in a project. Regarding my sample of projects, Table 15 indicates how GEF funds are allocated across the OPs, i.e., I omit figures on enabling activities and short-term measures.

The table shows that major conservation efforts are directed towards *forest ecosystems* ($2.3 billion), which also attract the highest amount of GEF support (36 percent of the total of grants). Approximately $1.1 billion is invested in *coastal, marine, and freshwater ecosystems.*[181] 21.5 percent of the total GEF grants are allocated to these OPs. A comparatively smaller amount is spent on the other OPs, namely arid and semi-arid ecosystems, mountain ecosystems, and the conservation and sustainable use of biological diversity important to agriculture. The share of the total GEF grants allocated to these programs ranges between 2 and 17 percent.

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[180] 137 projects (21 percent) are carried out in the LMMCs. The sum of the total project cost is $1.52 billion (21 percent). The GEF grant volume is $0.48 billion (28 percent).

[181] This refers to marine ecosystems within sovereign territories only, i.e., the high seas are not addressed in the GEF portfolio.

Table 15:
GEF Biodiversity Projects and their Allocation across Operational Programs

Operational program	Projects	Aggregate of total project cost (in million $)	% of total of GEF grants
Arid and semi-arid ecosystems	73	1,034.2	17.4
Coastal, marine, and fresh water ecosystems	99	1,132.7	21.5
Forest ecosystems	126	2,313.3	35.9
Mountain ecosystems	30	301.9	8.2
Agro-biodiversity	11	127.8	2.0

Source: GEF project database, own calculations.

The allocation of grants among countries is certainly influenced by factors other than biodiversity abundance. Socioeconomic factors can especially have an impact upon the allocation of funds (see sections 2.1 and 2.2). I briefly study this aspect by analyzing the allocation of funds with respect to two socioeconomic parameters.

First, the *per capita income of the recipient countries* is considered. Given the calls for the integration of conservation and sustainable economic development on a local project level, I ask whether such integration prevails on a national level, i.e., whether the GEF, whose task is to ensure environmental protection, primarily provides grants to the very poor recipient countries. For the analysis, I assign the country-specific gross national income (GNI) per capita (Atlas method; current dollars) of the recipient country to each individual project. For this purpose, I seek the income per capita level for the year of the project's approval. I could find this information for 600 of the 652 projects.[182] The aggregate cost of this subset of projects amounts to $4.03 billion.

I then classify the projects into four income classes that the World Bank suggests.[183] For each class, I calculate (1) the number of projects and (2) the aggregate grant amount for these projects. I depict the distribution of both variables in Table 16. It turns out that 41 percent of the projects considered are implemented in low income countries and 81 percent in countries belonging to the two lower classes. The third column illustrates that almost identical portions are derived with regard to the total GEF grants spent on these projects.

182 Cross-border projects are omitted, since the countries involved often belong to different income classes and, therefore, an income class cannot be assigned unambiguously to these projects.

183 The data on GNI per capita are taken from the WDI 2004 database (World Bank (2004)). For the classification, the income limits as presented for the 2004 classification are used.

Table 16:
GEF Biodiversity Projects and GEF Grants: Allocation across Countries According to Their National Income

Country	Number of projects (in recipient countries)		Aggregate of GEF grants (in million $)		Recipients as CBD parties (Number of countries)	
Low income	246	(41)	639.23	(42)	69	(41)
Lower middle income	241	(40)	624.58	(41)	54	(32)
Upper middle income	109	(18)	250.83	(17)	31	(18)
High income	4	(1)	2.83	(0)	15	(9)
Total	600	(100)	1,517.47	(100)	169	(100)

Note: Figures in parentheses indicate percentages.

Source: GEF project database, own calculations.

To evaluate the distribution, I again define a benchmark: for the 139 countries representing the community of recipients, I determine the mean GNI per capita (Atlas method; current dollars) from 1991 to 2003. The distribution of countries according to the income classes is indicated in the last column of the table. It turns out that this distribution resembles that of the other columns. This implies that the allocation of GEF projects and funds is neither biased towards the relatively wealthy recipient countries nor the very poor countries. It can be observed that the high income countries are slightly underrepresented in the allocation of projects grants, while the lower middle income countries are slightly overrepresented; they comprise only 32 percent of the recipients group but receive 41 percent of the total grants.

As a second socioeconomic parameter, I study the quality of *governance* in the recipient country. The hypothesis is that the GEF wants to ensure the resources it provides for conservation are used cost effectively. Poor governance in developing countries rich in biodiversity is considered an additional stimulus for biodiversity loss (Smith et al. 2003). Following this hypothesis, more GEF funds may be allocated to countries that have witnessed good governance in the past.

Two caveats need to be mentioned immediately. First, this hypothesis disregards that, in practice, recipient countries are not equally biodiversity abundant. More specifically, when some MDCs display substantial government failures, there is still reason to support conservation actions in these countries when they host exceptional biodiversity. From the point of view of the GEF, poor governance can lead to increasing costs of enforcement, which can be interpreted as an increase in the price of conservation. The second caveat refers to the fact that the quality of governance is difficult to assess. The indicators defined for this purpose are often the subject of debates (e.g., Kaufmann et al. 1999).

Taking these caveats into account, I perform a simple descriptive analysis of the hypothesis by using the indicators of the World Bank's *Country Policy and Institutional Assessment* (CPIA). The CPIA indicators summarize information concerning multiple economic and institutional parameters. The summarized data are used to classify recipient/developing countries into five quintiles. The first quintile (CPIA 1) describes the countries with the best governance and the last quintile (CPIA 5) those with the worst.

For the analysis, I use the CPIA classification of 2002 to 2004. Accordingly, I only consider GEF projects that were approved between 2001 and 2004. The CPIA rating of the corresponding recipient country is assigned to each of these projects.[184] Given the data available, this is done for 108 projects. The total amount of GEF grants to this subset of projects is $195.6 million. It is shown that the countries in the first and second quintile each attract 30 percent of these GEF funds. The percentages of countries in the mean quintiles are 14.5 percent for CPIA 3 and 15.5 percent for CPIA 4. Countries in the last quintile obtained 10 percent of the total amount of grants.

To conclude, although this simple analysis requires further refinement, it documents some evidence that the GEF tends to allocate more means to well-governed recipient countries than to countries with a poor governance structure. A further question is whether this result is the intention of the GEF policy or whether it is due to the fact that countries with bad governance are unable to meet the institutional requirements for obtaining GEF funding.

4.2.3 Transfers for Protected Area Projects: An Empirical Analysis

So far, I have analyzed biodiversity projects in an aggregate perspective; I have said little about the specific activities in the projects. Regarding the various levels of protection that can be implemented in critical ecosystems (see IUCN Guidelines; see also Section 4.1.2.2), ecosystem services as global public goods are either generated in more or less strictly protected ecosystems or provided as a joint product of sustainable resource management in protected landscapes.

Although GEF biodiversity projects aim at the conservation and sustainable use of conservation, not all of them necessarily make use of an approach of (strict) protection. In order to obtain a more precise picture regarding the use of *protected areas as a policy instrument*, the analysis should ideally only consider these projects if they aim at the establishment and management of such areas. Accordingly, the 652 projects have to be classified in this respect.

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184 For projects approved in 2001, I use the 2002 classification. For projects from 2002 onwards, I apply the data on overall rating.

In order to define a sample of projects for my further analysis, I adjust the sample of all projects by excluding the projects that do not predominately make use of *instruments other than the protection* of natural areas. At first, I exclude all enabling activity projects from the sample, since these projects assist the resource countries' national biodiversity policy in general, whereas the specific contribution to efforts in protected areas cannot be identified. In addition, I exclude all projects of the GEF operational program on "agro-ecosystems," since this program predominantly addresses the management of modified ecosystems. Finally, I classify the remaining projects on a case-by-case basis using information presented in the accessible *official GEF project documents*. These documents are published by the implementing agencies and are publicly accessible for the great majority of the projects.

As illustrated in the documents, the individual project typically addresses a bundle of issues and corresponding actions. To structure the different tasks, the actions are usually subsumed under four to five project components. Each component addresses one or several specific tasks in biodiversity conservation and sustainable use. A project is included in my study sample whenever protected areas are involved in at least one project component. Then the project is considered with its total project cost and total funding.[185] This procedure reduces the study sample to *295 projects*. The aggregate costs of these projects amount to *\$3.94 billion*. The GEF funding amounts to *\$1.38 billion*.

Given the adjusted sample of GEF funded protected area projects, I analyze the character and extent of the natural areas that have been proposed as project sites (see Section 4.2.3.1). Furthermore, I study the project actions that take place within these sites (see Section 4.2.3.2). Finally, based upon the information in the project documents, I describe the individual financiers of the projects in more detail (see Section 4.2.3.3).

4.2.3.1 Natural Areas as Project Sites

This section studies how the funding provided is invested, in particular what kind of natural areas is placed under reinforced or even newly established protection. The protected natural areas in the projects are categorized with respect to

[185] By taking into account the entire project, I also considered the entirety of actions in the projects. However, protected area measures may only constitute a fraction of the bundle of project actions. By including those actions that do not explicitly refer to protection, the extent of protected area measures in the GEF project portfolio is presumably overstated. Nevertheless, this procedure seems to be the only tractable way, since detailed information on the funding of components is often not available and since it is often hardly possible to distinguish components that refer to protected areas unambiguously from those components that do not.

economic criteria, such as property right regimes and types of human use (see Section 4.1.1.2).

Regarding land use and landownership in the project sites, the empirical evidence shows that various forms of property rights are involved. Marine ecosystems often represent common property resources with access regulations. In such cases, the GEF projects frequently aim at the change and improvement of the allocation of use rights, particularly with respect to *fishery*.

When considering terrestrial ecosystems, a lot of projects address *national protected area systems*, which consist of large parts of natural areas owned and administered by the public sector. The building blocks of such systems are national parks. These parks often encompass a core area that is strictly protected and a surrounding buffer zone where the local population is allowed to extract resources for subsistence. The GEF funding is often invested in activities in the context of park management.

Several other projects support government initiatives to protect *biodiversity on private land*. Examples are the Ecomarkets Project in Costa Rica or the project on Private Land Mechanisms for Biodiversity Conservation in Mexico. These projects implement mechanisms that provide payments to private landowners for the maintainance of vital ecosystem services. In Mexico, for example, these mechanisms are applied to establish conservation corridors on private land that connect state-owned strictly protected areas. Contracting for conservation on private lands is considered an appropriate instrument depending upon the conservation needs of certain endangered species or ecosystems and the extent to which private landowners can assist conservation or (already) foster biodiversity. Forms of community-based conservation are also chosen because they help to avoid conflicts on land tenures by maintaining the traditional property rights of indigenous people. This applies, for instance, to the project on the Indigenous Management of Natural Protected Areas in the Peruvian Amazon.

Generally, a GEF biodiversity project contains several components that address different forms of conservation actions. In this regard, project documents show that protected area projects often comprise actions in different systems of land property rights. This applies, for example, to the Colombian project on Conservation and Sustainable Use of Biodiversity in the Andes Region or the Jozani-Chwaka Bay National Park Development Project in Tanzania. Because of the mixture of the land property rights involved and their incomplete description in the documents, it is not possible to categorize the projects according to the types of landownership. Therefore, it also is not possible to derive the proportion of public land tenure compared to private or communal land tenure in the projects with sufficient accuracy.

The variety of project activities also impedes us in identifying the *degree of protection* to which the political stakeholders aspire and, thereby, the degree of human use that is excluded in the project sites. The ICUN categories of protected areas describe distinctive degrees of protection (IUCN 1994). Referring to these categories, James et al. (1999b) argue that in developing countries, strictly protected areas of category I are mostly unhabitated areas and that "exclusion is felt most acutely in II, III, and IV areas." In this context, evidence shows that some projects consider resettlement, such as in Benin or Uganda, which in some respect may indicate that a high level of protection is aspired to. In contrast, in other projects, such as the one in the Colombian Andean Region, resettlement is strictly avoided for political reasons. In the latter case, natural areas serve as multiple use protected areas or category V and VI areas, which allow for resource use by the local inhibitants.

Regarding the identified protected area projects, it is not feasible to unambigously assign IUCN categories of protection to each of them. This is not only because projects often combine actions of strict protection with community-based conservation activities but also because project sites often represent protected areas of different categories that are contiguous in the sense that one category nests within another (IUCN 1994).

Aside from land property rights and the degree of use exclusion, I investigate the *extent of the project sites*. It is of interest (1) how many natural areas are placed under some kind of protection as a result of GEF project actions and (2) whether the proposed project sites represent natural areas that are already protected (and, if so, what their spatial expanse is).

According to indications by the GEF (2004a), the projects that were approved between 1991 and 2003 address 1,232 (existing and newly established) protected areas, which together comprise an aggregate size of 256.75 million hectares. Regarding the individual projects, the information in project documents suggests that certain projects focus upon actions at a specific and clearly defined site, while other projects address a general program with a more broadly defined spatial scope.

Given the sample of 295 projects, I have found information on the size of the project area for 152 of them. In the specific case, the project area extends from less than 200 hectares for an island in Mauritius to more than 45 million hectares for sites in the Algerian desert. On average, the project actions address a total area of about 1.7 million hectares. Nevertheless, for more than half of the projects considered, the project area is smaller than 300 thousand hectares. Approximately 73 percent of the project sites identified cover less than one million hectares.

It is shown that in most cases, project sites have already been legally designated as protected areas before the project is implemented, i.e., the project actions support conservation and resource management on these sites. Nevertheless in some projects, the spatial extension of protection is planned. Evidence in this respect can be found for at least 61 projects.

It can be observed that *newly established protected areas* are a result of both increased protection on private lands and an increase in the land tenures of the public sector. For 27 projects, I identify empirical figures for the spatial expansion of national systems of protected areas. The extent of the expansion ranges from a few thousand hectares to 12.6 million hectares. The average expansion is 920 thousand hectares. In less than half of the projects, protection is expanded by less than 225 thousand hectares.

It turns out that the expansion of protection primarily relates to the acquisition of land by *public sector authorities*. There are typically two channels through which this acquisition occurs. On the one hand, the public sector purchases natural areas in private property in order to integrate them into the public protected area system;[186] on the other hand, unmanaged open access areas or "unclaimed government lands," such as in the Amazon Region Protected Areas Project in Brazil, are declared legally and actively managed as protected areas.[187]

Considering those projects where *land purchases* are planned, I investigate whether the acquisition of natural areas actually relies upon the international funding provided by the GEF. If this is the case, the GEF would directly serve as a mechanism that provides compensation for relinquished land development. Evidence in the documents suggests that, in practice, it is the domestic government or some local NGO that carries out the transaction of land property titles. However, the government/NGO may use the funds the GEF has granted for the land purchases. Alternatively, GEF funds may indirectly serve as compensation if they replace domestic funding in the resource management so that these funds can be invested in the land purchases.

To ascertain the impact of international financing, it needs to be known what actions governments and the private sector in the resource countries would have undertaken in the absence of any GEF support. Information on such a domestic benchmark relative to a GEF supported project alternative is provided in the incremental cost assessment in the project documents (see Section 4.3.3). I dis-

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[186] Evidence on land acquisition is, for instance, found for the Choco-Andean Corridor Project in Ecuador, the Biodiversity Conservation Project in Argentina, the Cape Peninsula Biodiversity Conservation Project in South Africa or the Protected Areas and Wildlife Conservation Project in Sri Lanka.

[187] An example from the Biodiversity Conservation Project in Boliva illustrates this point. On the project site, which represents a proposed national park, commercial use once took place but then was abandoned due to the lack of profitability: "A small part of the area has been divided into logging concessions, however, the isolation of the region, and the lack of access to high value trees has led to the area being abandoned by the timber companies" (see project document).

cover that in 25 projects, the extension of protected areas is an integral part of the GEF project alternative, i.e., the extension would not have been reinforced to the same extent in the domestic benchmark. For the remaining projects, the extension of protected areas is completed under the domestic benchmark or information for a distinction is not given.

Finally, for some projects, the documents indicate figures on the cost of land acquisition and the land prices. For example, for the Biodiversity Conservation Project in Argentina, the documents provide figures on the extension of a protected area, as well as expenditures for the area acquisition. The ratio yields a price of $4.7 per hectare. Similarly, the documents for the Sustainable Protected Area Development in Namaqualand Project in South Africa show a (calculated) gross land price of $17.95 per hectare, which, however, includes the costs of zoning. Futhermore, in another Argentinian project named Management and Conservation of Wetland Biodiversity in the Esteros de Ibera, private landholdings are acquired at a price of $50 to $100 per hectare to establish protected areas. These examples imply that the stakeholders in the project planning apparently consider marginal and less productive land for the extension of the protected area systems.

4.2.3.2 Protection Measures as Project Actions

The previous analysis implies that only a minor part of GEF funding is used as direct compensation for forgone revenues from local land use other than conservation. I therefore ask what activities then receive the financing. For this purpose, I study the information provided in the *project checklists* that are contained in some of the official documents. A project checklist serves as a pattern to categorize the proposed actions. According to the GEF guidelines on project proposals, the checklist is a nonmandatory but recommended element of the project brief document for medium-size projects. I found it for 38 protected area projects. In addition, four full-size projects on protected areas contain project categorization sheets that have a similar structure to the checklist and are therefore included in my analysis.

In the checklist, the measures applied are classified in eight project activity categories plus seven technical categories. Given the data for the 42 projects, I study how frequently each category applies. Multiple categories can apply in each project. Considering first the *project activity categories*, it turns out that, aside from protected area zoning/management, inventory/monitoring is most frequently addressed (88 percent of all projects). In addition, ecotourism (74 percent), buffer zone development (64 percent), and benefit sharing are also often agreed upon measures. In contrast, agro-biodiversity activities (43 percent) and trust funds (14 percent) are addressed in less than half of the projects.

Turning to the *technical categories*, it is shown that awareness/information/training is addressed in nearly every project. Further technical support focuses upon institutional building (93 percent) and technical/management advice (91 percent). Furthermore, targeted research is applied in 57 percent of the projects. In contrast, technology transfer (45 percent) and investment (41 percent) are part of the actions in a fewer number of projects.

Considering the various entries in activity categories and technical categories, each project displays an individual profile. I compare the various profiles across the projects in order to identify potential complementary or substitution relationships between the various categories, i.e., applied actions. For this purpose, I represent the binary checklist entries for each category as a 42×1 array and calculate the Pearson correlation coefficient between each pair of categories. The matrix of the coefficients is presented in Table 17.[188] The results of the correlation are used to suggest implications of the relationship between the categories.

It turns out that, in most cases, the correlation is quite low. Consequently, there is obviously no unique pattern for actions in the projects considered. Moreover, the establishment and management of protected areas in the GEF projects seems to demand a bundle of measures whose composition depends upon the specific ecological and socioeconomic environment of the project site.

More specifically, any pronounced correlation might provide evidence of potential complementary or substitution relationships between the categories: *complementary relationships* are shown for several combinations, in particular, for institution building and inventory/monitoring. Less pronounced complementary relationships are, for instance, shown for buffer zone development and protected area zoning/management, policy advice and technology transfer, or for policy advice and institution building. Furthermore, it turns out that trust funds are weakly complementary to benefit sharing, as is investment to target research. Finally, *substitution relationships* are indicated for trust funds and awareness/information/training, trust funds and agro-biodiversity, as well as for investment and agro-biodiversity.

To resume, these results only provide little evidence of potential general technological relationships of protection. In addition, the sample of checklists predominately refers to medium-size projects, which effectively reverses the proportion of medium-size to full-size projects that is observed in the study sample. Full-size projects are typically more extensive and require more financial resources. In this respect, the findings from the project checklists may be biased against actions that include large-scale investments and that, therefore, cannot be part of a medium-size project.

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188 Statistical testing with the calculated coefficients cannot be performed, since the checklist entries do not represent normally distributed variables.

Table 17:
Matrix of Pearson Correlation Coefficients for Entries in Project Checklists

Correlation (42 projects)	PA zoning and Management	Buffer zone development	Inventory/monitoring	Ecotourism	Agro-biodiversity	Trust funds	Benefit sharing	Other activities	Institution building	Investment	Policy advice	Targeted research	Techn./mgt. advice	Technology transfer	Awareness./info./training	Other tech. categories
PA zoning and mgt.	1.00															
Buffer zone development	0.37	1.00														
Inventory/monitoring	–0.10	–0.27	1.00													
Ecotourism	0.26	0.23	0.12	1.00												
Agro-biodiversity	0.24	0.04	0.32	0.08	1.00											
Trust funds	0.11	0.02	–0.06	0.09	–0.35	1.00										
Benefit sharing	0.15	–0.01	0.15	0.28	0.13	0.34	1.00									
Other activities	–0.01	0.28	–0.23	0.28	–0.16	0.02	0.03	1.00								
Institut. building	–0.06	–0.17	0.69	0.12	0.20	0.09	0.27	–0.35	1.00							
Investment	–0.14	0.13	–0.29	0.17	–0.35	0.14	0.11	0.22	–0.28	1.00						
Policy advice	–0.18	0.08	0.26	0.10	0.12	–0.04	0.12	0.08	0.35	0.11	1.00					
Targeted research	–0.24	–0.14	0.28	0.14	–0.13	0.22	0.17	–0.04	0.27	0.35	0.20	1.00				
Techn./Mgt. advice	0.22	0.10	–0.12	–0.01	0.12	0.13	–0.27	0.04	–0.07	–0.06	–0.03	–0.12	1.00			
Techn. transfer	0.07	0.08	0.33	0.11	0.28	0.18	0.26	0.01	0.21	0.05	0.36	0.01	0.13	1.00		
Aware./info./training	–0.04	–0.12	0.42	–0.09	0.14	–0.38	–0.13	0.10	–0.04	–0.19	0.25	–0.14	–0.05	0.14	1.00	
Other tech. categories	0.12	0.33	–0.03	0.27	0.13	0.00	0.11	0.25	0.09	0.21	0.00	0.00	0.15	0.24	0.07	1.00

Source: Selected GEF project documents, own calculations.

4.2.3.3 GEF Funding Compared to Funding from Other Sources

Returning to the financing of the project actions, the analysis in Section 4.2.2 provides only very little information on how precisely the funds have been raised aside from the resources provided by the GEF. In this regard, the official project document provides more detailed information. For 259 of the 295 protected area projects, the documents describe the financiers of the project budget in a complete and consistent manner. The aggregate costs of these projects amount to \$3.51 billion. The total of the GEF grants is \$1.22 billion.

In order to describe the different financiers, I classify them into six groups. In addition to the GEF, these are

- *other multilateral donors*, for example the UNDP, the World Bank, the United Nations Foundation, the Ramsar Small Grant Fund (see Section 4.1.2.1),
- *official unilateral donors*, i.e., mainly agencies of development assistence in the developed countries, and
- *domestic public sector institutions* in the resource countries, i.e., for example, the Ministry of the Environment or domestic public universities that are engaged in biodiversity research.

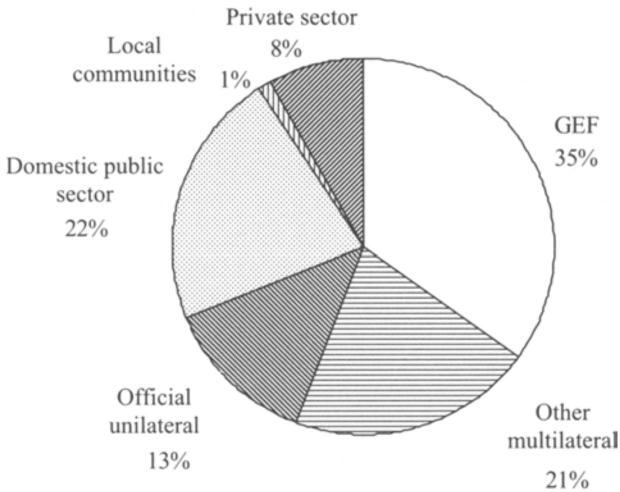
Finally, the *private sector* participates in the project financing. This includes

- local communities and nongovernmental stakeholders at the projects sites, and
- domestic/foreign firms, foundations, and NGOs that provide funding for non-profit activities on biodiversity conservation.

Based upon this classification, I assign the indicated contributions to the project budget to the different financier classes. This is done for 253 projects. Sometimes funding is provided through collaboration between stakeholders who belong to different classes. For instance, a university in a donor country collaborates with a university in a recipient country, or an official donor, such as the World Bank, collaborates with a local NGO or municipality on the project site. Since, in such cases, the documents do not report how the funding is broken down with respect to financiers in the different classes, I fully assign the indicated contributions to the class that appears to be the primary provider of the funding. After breaking down the funds in all of the projects and assigning them to the different classes, I derive both (1) the shares in the coverage of the aggregate project costs and (2) the average cost coverage for each class of financiers.

The *shares* on the *aggregate project costs* are illustrated in Figure 20. It shows that the GEF covers more than one third of the aggregate project costs (35 percent). The other multilateral and unilateral donors together provide resources for about another third, so that *international donors* together cover nearly 70 percent of the costs incurred in the biodiversity projects considered.

Figure 20:
Funding of GEF Protected Area Policies by Financiers



Source: GEF project database (selected GEF project documents), own calculations.

The *resource countries* participate in the cost coverage through the contributions of public sector insitutions (22 percent) and local communities (1 percent). Contributions from the private sector (excluding domestic local communities) cover about 8 percent of the aggregate project costs. These resources include revenues from ecotourism within the project sites. However, such forms of *self-financing* protection activities are only provided for in a few projects (see Section 4.2.4). Regarding the remaining nonprofit actions of the private sector, domestic and foreign donors often cooperate, as shown by collaborations between domestic *international NGOs* and their *local partners* located in the resource country. Although anecdotal evidence in the project documents suggests that the international donors provide the larger part of the funds in this class, the share of this class cannot be unambiguously assigned to either the domestic or international sphere.

The average figures constitute a large share of total funding from international donors (at least 68 percent). This may suggest that substantial cross-border and *global externalities* are generated in these projects. However, it needs to be mentioned that a part of the official multilateral funding is provided on a loan basis, i.e., these funds have to be repaid by the resource country. Against this background, the described funding shares for domestic financiers and international donors have to be qualified; the actual shares of the resource country is apparently larger.

When turning to the domestic financiers and taking the potential international loans for funding domestic biodiversity conservation into account, one may ask whether these figures represent the *incremental benefits* to the resource country that the GEF-supported conservation activities generate in addition to the global external benefits. The answer to this question involves using the incremental cost principle, which Section 4.3 studies in more detail. In both respects, a straightforward inference from the funding shares to the extent of domestic benefits and international externalities is likely to be misleading.

For reasons of completeness, I use the data on the financing of the individual project to derive the *average coverage* of *total project cost* for each class of financiers. The average cost coverage by the GEF (49 percent) is above the share of the coverage of the aggregate costs (35 percent) described in the figure. This implies that the GEF plays an important role in the funding of small and medium-size projects (see Section 4.2.2). In contrast, the domestic public sector of the resource country, on average, covers 18 percent of the costs, which indicates its strong participation in large projects. The other mean shares are 11 percent and 9 percent for official multilateral and unilateral assistance, 1 percent for local communities, and 12 percent for the remaining private sector.

Finally, some general remarks have to be made. First, with respect to international funding in GEF projects, it is often argued that the participation of the GEF *leverages additional funds* from other unilateral and multilateral donors, i.e., GEF funding catalyzes extra funding from international sources. The presented results cannot be used to confirm this assertion, since the figures in the official documents on the project funding do not distinguish payments that are somehow associated with the GEF grant from payments that are provided regardless of the GEF assistance.

Furthermore, the empirical findings relate specifically to protected area projects. They may not be generalized for all GEF biodiversity projects. Because of an incomplete breakdown of the project actions and the corresponding costs and funding, I consider the projects with the complete set of actions and the associated funding. As a result, the extent of the GEF portfolio on protected areas is likely to be overstated.

GEF Biodiversity Projects on the Preservation of Genetic Diversity

Before proceeding with the analysis of protected areas, I should mention that the GEF biodiversity portfolio also adresses the preservation of genetic diversity, which I investigated in Chapter 3. As argued in that context, the preservation and use of genetic resources, as well as of specific indigenous knowledge, displays

Table 18:
Selected GEF Biodiversity Projects Dealing with the Preservation of Genetic Diversity and Indigenous Knowledge

Resource country	GEF project	Year of approval	Total project cost (in million $)
Jordan	Conservation of Medicinal and Herbal Plants	2002	12.85
Regional	In-Situ Conservation of Crop Wild Relatives through Enhanced Information Management and Field Application	2002	12.68
Vietnam	In-Situ Conservation of Native Landraces and Their Wild Relatives in Vietnam	2002	3.92
Regional	Conservation of Gramineae and Associated Arthropods for Sustainable Agricultural Development in Africa	2002	2.54
Ecuador	Albarradas in Coastal Ecuador: Rescuing Ancient Knowledge on Sustainable Use of Biodiversity	2001	3.11
Zimbabwe	Conservation and Sustainable Use of Traditional Medicinal Plants	2001	1.63
Regional	Community-Based Management of On-Farm Plant Genetic Resources in Arid and Semi-Arid Areas of Sub-Saharan Africa	2001	2.05
Regional	Biodiversity Conservation and Integration of Traditional Knowledge on Medicinal Plants in National Primary Health Care Policy in Central America and Caribbean	2001	1.55
Egypt	Conservation and Sustainable Use of Medicinal Plants in Arid and Semi-Arid Ecosystems	2000	9.05
Peru	In-Situ Conservation of Native Cultivars and Their Wild Relatives	1999	6.42
Ethiopia	Conservation and Sustainable Use of Medicinal Plants	1999	6.81
Regional (Mid East)	Conservation and Sustainable Use of Dryland Agro-Biodiversity of the Fertile Crescent	1998	18.53
Regional	Participatory Management of Plant Genetic Resources in Oases of the Maghreb	1998	6.58

Source: GEF project database.

some public good properties. It is, therefore, not surprising that several GEF projects have been approved that deal with this issue. To illustrate, Table 18 presents selected projects in this regard. This collection of projects is based upon indications in the project name. I do not claim completeness, since further projects may include actions on genetic diversity in project components.

Suggestive evidence implies that the preservation of genetic diversity for agricultural production is dominant in the selected projects. Nevertheless, certain projects also address medicinal plants as part of genetic resources in the health care sector. Several projects that address conservation for this sector combine conservation with traditional and community-based use.

4.2.4 Sustainable Finance of Protected Areas in GEF Projects

Protected areas represent a means to ensure that the flow of the ecosystem services with a global public good character is maintained. The maintenance of this flow over an extended period demands the *sustainable finance* of an effective protected area management (McNeely 1997). In this regard, the financing in the form of (medium-term) projects that carry out one-time capital investments may hardly be sufficient. Additional or other financing mechanisms are required. In the following, I (1) describe which mechanisms are, in principle, available. By focusing upon ecotourism and environmental funds, I (2) ask how these two mechanisms are applied in the GEF projects. This relates to the general question as to whether and how the projects address financial sustainability and, thus, guarantee long-term effectiveness.

Another mechanism that is closer to realization is the linkage of conservation finance to carbon credit schemes in connection with international emissions trading schemes. Based upon regulations in climate policy on Joint Implementation and the Clean Development Mechanism, developed countries finance projects in transformation and developing countries that provide for emissions avoidance (mitigation) in the context of land use, land use change, and forestry. Projects on afforestation or reforestation in the developing countries jointly contribute to biodiversity conservation in particular (Clemençon 2000; McNeely 1997).[189]

Independent of the geographical scope of the external benefits of ecosystem services and the land tenures on the considered ecosystems, two mechanisms have attracted increasing attention as an instrument for sustainably financing protected areas. These are financing through

- ecotourism,
- environmental funds.

4.2.4.1 Revenues from Ecotourism

The term *ecotourism* is used for nature-based tourism (visitation) that is consistent with sustainable ecosystem use (Lindberg 2003). General aspects of ecotourism services as an economic good have been addressed in the context of market creation for ecosystem services in Section 2.3.3. These services are localized ecosystem services that are, to some extent, nonrival in consumption

[189] The implementation of these mechanisms is still in progress. It needs to be studied as to whether actions on carbon sequestration, afforestation, and reforestation are indeed complementary to biodiversity conservation and whether financial sustainability of conservation measures is ensured.

but can be made exclusive. Access to tourist sites can be restricted and an entrance fee can be levied. In this regard, ecotourism services represent impure public goods or club goods.

Similar to the preservation of in situ genetic diversity, ecotourism sites, such as national parks, natural monuments, and protected landscapes, may jointly provide ecosystem services that have the character of a local or global public good. Examples of *joint products* are supportive services that generate indirect/primary values on a local level and cultural services that provide nonuse and option values on a global level (see Section 2.3.2).

Ecotourism as a means to finance and support conservation can be less effective for two reasons. First, the profitability of ecotourism may trade off with its *compatibility with conservation*: in order to raise sufficient profits and, thus, create incentives for conservation, ecotourism frequently demands continuous and potentially large-scale tourist visits to the natural sites. In many cases, this has adverse ecological and social impacts and is therefore not compatible with sustainability. Moreover, in such cases, the supply of tourism services does not contribute to the generation of public ecosystem services, but, rather, leads to biodiversity degradation (Brown 1996; Huybers and Bennett 2002; Vaughan 2000).

Considering the social conflicts arising from ecotourism, domestic governments and local stakeholders may have diverging interests with respect to profitability. While central governments often regard ecotourism as a source of revenues that supports overall economic development, municipalities and local communities prefer to integrate ecotourism into the local and community-based development process (Naidoo and Adamowicz 2005; Wunder 2000).

Second, even if biodiversity-friendly and socially accepted uses of ecotourism are implemented, it is unclear as to whether the revenues obtained are sufficient to enable sustainable financing of tourism infrastructure, including protected area management. The question of *financial sufficiency* relates to both to the current size of tourism revenues and the future development of world markets for these ecotourism services. When protected area management is dependent upon tourism revenues, fluctuations in the market demand for these services cause fluctuations in the financial endowment that is available for the management units and, hence, for the actions that the units can undertake. For example, a drop in tourism revenues can lead to diminished quality of the flow of ecosystem services generated at tourist sites. Due to ecological irreversibilities, it may be impossible to restore original ecosystem quality, even if the market revenues from ecotourism potentially recover over time.

Empirical evidence of the attainable market revenues is often derived from figures on *international tourist receipts*. Data by the World Tourist Organization shows that the value of receipts worldwide is currently approximately $524.2

billion.[190] In the recent decade, this value (in current dollars) has increased, on average, by about 5 percent per annum. However, only 20 percent of this occurs in the developing countries of Latin America, Africa, and Asia.[191] Furthermore, according to the Ecotourism Society, 40 to 60 percent of the worldwide tourism income can be attributed to nature-based tourism (Vaughan 2000). This would correspond to a market of $44 to $65 billion, which is distributed among the developing countries. Empirical data on a country level show that certain countries, such as Mexico, Thailand, and Malaysia, capture a substantial market share, while others, particularly in Africa, do not manage to participate in the market.

Regarding the site level, Lindberg (2003) finds anecdotal evidence that revenues from tourism contribute noticeably to the extent of protected areas and the quality of management. He, however, notes that not all revenues from tourism actually go to conservation management but that these revenues typically are not the only funding source for the management.

To resume, ecotourism may indeed be a helpful tool to finance conservation. However, its effectiveness has to be evaluated in the site-specific context.

GEF Projects on Protected Areas and Ecotourism

Turning to the GEF biodiversity projects, I investigate the role that is assigned to ecotourism as a mechanism to finance protected areas. For this purpose, I draw partly upon the results from the previous analyses in Section 4.2. The study of entries in project checklists reveals that ecotourism is frequently part of the activities in medium-size projects: 31 of the selected 42 medium-size projects display an entry for ecotourism. Nevertheless, I cannot confirm this share for the overall sample of protected areas projects, since the project checklists for full-size projects are underrepresented in my analysis (see Section 4.2.3.2).

Furthermore, in constrast to the actions on the preservation of genetic diversity presented in the last section, nature-based tourism is a major subject in the projects. More specifically, ecotourism is often included in project components that aim at the *integration of conservation and community-based development* within or in the proximity of protected areas.

In the context of the project financing analyzed in Section 4.2.3.3, it is shown that tourism income is only a source of funding project actions in a few projects (*self-financing*). Among these cases, I find mixed results regarding the extent to which ecotourism revenues cover the total project cost: for example, for the Hon

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190 The data is presented at http://www.world-tourism.org/facts/menu.html. The data cited here were downloaded on June 15, 2005.

191 The shares derived are based upon my own calculations using data provided by the World Tourist Organization for 2003. Hong Kong and Singapore are not included in the group of developing countries.

Mun Marine Protected Area Pilot Project in Vietnam (3 percent of the total project costs) and the Esteros del Ibera Project in Argentina (2 percent), the documents indicate only a small share of tourism income relative to the project costs. In contrast, ecotourism contributes more substantially to the cost coverage in the Komodo National Park Collaborative Management Initiative in Indonesia (40 percent) or the Eg-Uur Watershed project in Mongolia (21 percent).

However, these figures on self-financing by ecotourism refer only to the coverage of costs that accrue within the duration of the project. Regarding the sample of protected areas projects, it is shown that they extend over a 2 to 10 year period. Evidence of a follow-up project with GEF funding is only given for a few projects. In most cases, it remains unclear as to whether the project sites will be able to attract international funding in the future.

However, the temporary constraints of the GEF projects can contribute to the sustainable financing of protected area management if the project funding is invested in a way that a substantial stream of financial resources is facilitated in the future. *Investments*, in this respect, need to address infrastructure or productive resources that support the generation of market revenues from protected areas that can thus substitute for the international funding in the long run. Financial sustainability in this context requires that (1) sufficient project funds are used for investment purposes and (2) the investments provide sufficient returns, i.e., substantial (market) revenues are generated from the newly established infrastructure.[192]

To answer these two questions satisfactorily, my analysis requires extensive site-specific information. On a general basis, i.e., independently of the total size of the budget and the financial needs at a specific site, I investigate how many project funds are allocated to investments compared to noninvestment activities. For some projects, the official documents provide information in this regard; for 43 projects, the total project cost is broken down into investment costs and recurrent costs.

According to these documents, *investment costs* comprise costs for infrastructure development, technical assistance, vehicles, and equipment, as well as for training, civil work, research and surveying, and costs that accrue in the context of international cooperation (expenditures for regional meetings, travel, and consultations). In contrast, *recurrent costs* represent the costs accrued due to incremental salaries, vehicle operations and maintenance, and subsistence and travel allowances.[193] Based upon the data given for this subset of the projects, I

[192] As shown in the checklist analysis, investment as a technical category applies only in 17 of the 42 medium-size projects.

[193] In these examples, investment costs seem to refer to investments in human as well as in physical capital, while in the checklists, investments obviously relate primarily to physical capital.

calculate the share of investment costs in the total project cost. It ranges widely from 22 to 100 percent. The average share of investments is 80 percent.

Since the projects considered consist of several components that address various tasks of biodiversity conservation, not all investments that underly the derived figures refer to protected area measures. More detailed information in this regard is provided for four projects whose official documents describe investment and recurrent costs not only on a project level but also with respect to the individual project components. Given this data, it is possible to describe the share of investments for project components that directly address the management of protected areas: for the El Kala National Park Project in Algeria, the Biodiversity Protection Project in Belarus, and the Coastal and Biodiversity Management Project in Guinea-Bissau, the shares of investment costs are quite high (87 percent, 93 percent and 88 percent, respectively). In contrast, in the Transfrontier Conservation Areas Pilot Project in Mozambique, investment costs are only 57 percent.

To conclude, these data indicate that projects obviously primarily address investments. This serves as suggestive evidence that financial sustainability is taken into account in the GEF projects. However, it remains unclear as to to what extent sustainability is indeed attained in practice.

4.2.4.2 Environmental Funds

Resource management, which requires regular management input over a long period, demands suitable financing mechanisms. Environmental funds are considered a useful tool is this regard. In particular, they can assist the financing of the recurrent cost of protected area management (Mitkin 1995).

Several sources, including grants from international donors, revenues from domestic taxes, fees, and private donations, may provide capital income for an environmental fund (Lambert 2003). While the fund's resources are used to finance conservation, the disbursement of the resources is structured differently across the various types of funds. In essence, three forms are identified: (1) an *endowment fund* maintains its capital stock over time and only spends the capital proceeds. In contrast, (2) a cash fund, or, synonymously, *sinking fund*, runs over a relatively long but fixed period, where both the original capital income and the proceeds are completely disbursed at the end. Finally, (3) a *revolving fund* rebuilds its capital income, which is decimated due to disbursements by regular contributions from the different funding sources (Lambert 2003; Smith 2000).

To safeguard protected area management in the long run, endowment funds are considered appropriate. For medium-term actions that terminate, are handed over to build-up organizations, or that develop other sources of recurrent funding, sinking funds are a useful tool. Overall, environmental funds disburse small or (at best) medium-size amounts. Accordingly, environmental funds are less suitable if natural resources face urgent threats that require a substantial funding response in the short run (Smith 2000).

Considering the *scope of actions* supported by grants from these funds, some (large-scale) funds are established on a national level and support the full portfolio of domestic conservation actions, while other funds focus upon the specific protected areas or the habitats of endangered species. Furthermore, funds are sometimes part of a hierarchical financing structure and provide grants to other groups or organizations engaged in biodiversity conservation (Lambert 2003).

The income the donors grant to the environmental funds represents tied capital. With respect to property rights to this capital, most funds are arranged as *trust funds*, i.e., a person or group (trustee) manages the capital stock according to predefined objectives and for the benefit of another person or group (Mitkin 1995). Regarding the *governance structure* of the funds in detail, most of them are established as private organizations. The board of the funds is either composed of both governmental and nongovernmental representatives or run by NGOs. Alternatively, a fund can be associated with a government agency (Lambert 2003).

Some studies analyze whether and under what conditions environmental funds lead to positive impacts upon conservation. The studies discover that funds with mixed boards demonstrate the most promising results, since this structure enables both active government support and the *participation* and collaboration of *people from different sectors*. Governmental assistance usually aims at the establishment of a stable legal framework in order to back political and administrative arrangements and provide technical support. The contributions by NGOs compensate for the drawbacks of a purely governmental approach, namely the avoidance of bureaucratic procedures and administrative bottlenecks that prevent the funds from reaching field-level activities. In addition, NGOs can provide information and contact to remote and small local communities that are not typically addressed by official programs. Furthermore, the remaining private sector contributes to efficiency, since the participating private actors are often experienced in serving on boards and offer the financial expertise that is required for the management of the trust fund (GEF 1998a; Lambert 2003).

Overall, a well-designed and corporate-governance-like structure of environmental funds enables a system of financial management and controls that supports transparency and accountability to contributors and other stakeholders. It avoids the opportunistic behavior of stakeholders (corruption, political discrimination) and reduces transaction costs. The studies argue that these advantages, in many cases, outweigh the opportunity costs of tying up capital (GEF 1998a; Lambert 2003; Mitkin 1995).

Finally, the optimum performance of an environmental fund also depends upon whether conservation is effectively enforced at the project sites receiving financial support. This requires monitoring and data collection, as well as measures that raise awareness and provide environmental education or offer management training to support local groups. In this respect, the GEF (1998a) concludes that a successful environmental fund is "more than just a financial mechanism."

Currently, many developing countries whose public sector only provides insufficient financial means for biodiversity protection and protected area management have implemented conservation trust funds. According to Quintela et al. (2004), over 100 environmental funds have been created over the last 15 years and most of them are still operating. These examples combine different trust fund models for different purposes in the context of conservation. Regarding the capital stock of these funds, anecdotal evidence shows that some large national funds amount to between \$20 and \$50 million.

GEF Assistance for Environmental Funds on Biodiversity Conservation and Protected Areas

Given the fact that environmental funds are financed by various sources, the GEF is considered the largest donor for environmental funds worldwide. The GEF assistance in this respect is allocated across (1) investment funds, (2) a small grant program, and (3) conservation trust funds.

GEF investment funds are designed to assist private sector activities that contribute to sustainable development through innovation, technology transfer, or resource mobilization. In the GEF's biodiversity focal area, the intention was to establish the Terra Capital Biodiversity Enterprise Fund in order to allow for equity and quasi-equity investments in Latin American companies that sustainably use or protect biodiversity (GEF 1998b). The *GEF Small Grant Programme* in association with the UNDP provides funding for NGOs and community-based organizations to enforce activities in the context of sustainable livelihoods. Sometimes the program collaborates with domestic environmental funds (Timpson 2000).

Regarding the *conservation trust funds* addressed in the GEF projects, three types of funds can be distinguished: first, support is given to *park funds*, which more or less aim directly at the management of protected areas. Examples in this regard are the Fund for Natural Areas Protected by the State (FONANPE) in Peru and the Table Mountain Fund (TMF) in South Africa. The GEF supports the former with a \$5.2 million grant. The latter receives a \$5 million grant. The second type is *grant funds*, i.e., funds that provide financial resources to domestic (small) conservation projects that cover a wide range of activities and, in particular, address actions by the private sector and/or local communities. The

Brazilian Biodiversity Fund (FUNBIO) is an example of this type of fund. Originally designed as a sinking fund, FUNBIO receives \$20 million from the GEF in addition to a \$10 million grant from the Brazilian Government. Finally, the GEF also assists conservation trust funds that combine elements of the two types of funds. Funding in this respect is granted, for example, to the Mgahinga and Bwindi Impenetrable Forest Conservation Trust in Uganda (\$4.3 million) or the Bhutan Conservation Trust (\$10 million) (Mitkin 1995; GEF 1999a: Annex C).

Overall, these examples of GEF support amounting to several million dollars are not representative. Given the information reported in the GEF (1999a, 1999b), only seven conservation trust funds are endowed with substantial funding by the GEF. In addition to the aforementioned funds, these are the Central American Fund for Environment and Development (FOCADES) and the Mexican Fund for Conservation and Nature, which receive \$15 million and \$16.5 million from the GEF. As suggested by the empirical evidence, the GEF frequently finances the preparation activities necessary for the creation and implementation of such funds. The capital provided for these funds is then generated from other sources (Mitkin 1995; GEF 1999b).

The public interest in environmental funds as mechanisms to provide sustainable finance rose considerably in the late 1990s when most of the (comparative) analyses on funds were carried out. Accordingly, the studies available only provide data on funds for that period. However, Quintela et al. (2004) argue that the environmental funds earmarked for conservation and protected area management still proliferate. In total, the GEF has so far supported the creation of 23 of them. In Africa, GEF allocations amount to a total of \$25.6 million.

To resume, multiple fund models exist in order to satisfy the need for protected area management at specific sites. Regarding the sufficiency and sustainability of the funding provided by these funds, sinking funds (and revolving funds) show a disbursement that is often larger than that of endowment funds. However, in contrast to the latter, they require new capital input from time to time. The extent to which the disbursements and annual capital proceeds are sufficient to finance the recurrent tasks of protected area management depends upon the site-specific costs, the capital that has been mobilized for the fund, and the capital return.[194]

The GEF plays various roles in the policy on environmental funds. In some cases, it participates in the capitalization of the fund; in others, it merely covers the transaction costs for its implementation. Furthermore, although several GEF projects on protected areas are linked to environmental funds, they are not a

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[194] Last but not least, the effectiveness of environmental funds depends upon whether the transfer recipient complies with the agreed upon terms on conservation measures or behaves opportunistically (moral hazard).

regular element of the projects. In many cases, the sustainable financing of protected areas depends upon the other domestic and international financing mechanisms.

4.3 GEF Transfers and the Incremental Cost Principle: Cost-Effectiveness and Incentive Compatibility

The study of the GEF biodiversity policy so far has concentrated upon positive issues regarding the allocation of funds. In order to conceptualize the GEF policy from a normative point of view, it is helpful to draw upon the findings on international transfers presented in Section 4.1.1.3. Based upon the idea of cooperation between sovereign countries, the literature offers three alternative explanations as to why there are transfer payments like those of the GEF (Cervigni and Pearce 1995). The first two explicitly rest on *moral grounds*:

- The developed countries are responsible for the protection of global biodiversity because they are comparatively wealthy. Since they depend upon the cooperation of the less wealthy resource countries, they offer transfers in order to satisfy the participation constraints of the resource countries.
- Alternatively, the protection of biodiversity may generate primarily domestic benefits. Nevertheless, resource countries are unable to finance the appropriate resource management due to constraints in available economic endowments. Since these constraints can be attributed to past policies of uncompensated resource exploitation by developed countries, it is justified that the latter now provide funding for sustainable resource use in the developing world.

The third explanation considers biodiversity as a *global common*, i.e., the benefits that are generated from its conservation are nonexcludable, while there are simultaneously competing uses for natural land as a biodiversity habitat. In contrast to the previous explanations, the emphasis falls upon the benefits from conservation, which are shared among the countries:

- Since benefits are nonexcludable, the extent of conservation in the noncooperative outcome with uncoordinated unilateral actions falls short of the Pareto-efficient level. International transfers are supplied to countries that either derive few benefits from conservation (and thus only invest few resources in the conservation of the habitats within their territory) or only possess limited resources and are, therefore, unable to assist conservation. The aim of (conditional) transfers in this regard is to *create incentives* for these countries *to cooperate.*

According to Cervigni and Pearce (1995), the latter point of view dominates the understanding in international biodiversity policies, including the GEF policies. The two major questions arise in this context. They refer to

- the cost-effective use of the GEF funding that donor countries provide to the GEF in order to support conservation in the developing world, and
- the sharing of the benefits generated from the GEF projects.

The latter question on *equity* is also of importance to the *efficiency* of transfer payments, since the sharing of the surplus from international cooperation can be the subject of political conflict that may inhibit a (timely) cooperative solution (Mohr 1990; Sandler 1993; see also Section 4.1.3). In order to meet the participation constraint of the resource country and render any strategic behavior on its part unprofitable, transfer payments need to be designed in an incentive-compatible way. The key question is how to account for extra domestic benefits generated from internationally supported conservation actions in the transfer policy. Since, from the point of view of the donor community, the cost-effective use of its financial resources demands complete accounting, the principles of *incentive compatibility* and *cost-effectiveness* trade off to some extent.

In practice, the GEF handles this trade-off according to the *incremental cost principle*, which guides the GEF project financing. In the following, I summarize the public understanding of this principle (Section 4.3.1). Based upon this, I review analytical models that discuss international transfer payments (and the incremental cost principle) from a theoretical perspective (Section 4.3.2). In Section 4.3.3, I perform an empirical analysis of incremental cost assessments in those GEF protected area projects identified in Section 4.2.3.

4.3.1 GEF Biodiversity Policy and the Incremental Cost Principle

Grant payments in the GEF biodiversity focal area relate to the formal relationship between the GEF and the CBD established in international law (see Section 4.2.1). Provisions in the CBD guide GEF payments to resource countries for the purpose of biodiversity conservation. More specifically, this is the *incremental cost principle*, as defined in CBD Art. 20(2).[195] These CBD provisions support the following points:

[195] The complete wording of CBD Art. 20(2) says that "the developed country Parties shall provide new and additional financial resources to enable developing country Parties to meet the agreed full incremental costs to them of implementing measures which fulfill the obligations of this Convention and to benefit from its provisions and which costs are agreed between a developing country Party and the institutional structure referred to in Article 21, in accordance with policy, strategy, programme

- Developing countries that have joined the CBD are committed to conserve their natural resources or manage them in a sustainable way in order to meet the Convention's objectives.
- The actions arising from this obligation and that the resource countries enforce within their territories have to be supported financially by the developed countries that are signatories of the CBD. The funding provided for this purpose should not result from a redirection of the flow of those financial resources that have previously been allocated to development assistance, but should represent new and additional resources.
- Considering each conservation activity or project implemented in this respect, the financial support from the GEF should correspond to the level of the "*full agreed incremental cost*" of the activity.
- While financial support is related to activities enforced in accordance with the Convention's objectives, it is explicitly stressed that the support should enable the developing countries to benefit from the Convention's provisions, i.e., the conservation and sustainable use of biodiversity.
- In order to facilitate financial support, the signatory countries of the CBD are called on to implement an institutional arrangement. In this regard, the signatory countries have appointed the GEF as the relevant funding institution.

In essence, these provisions describe that the incremental cost principle is an instrument to determine the amount of GEF funding to be granted to the resource countries in order to implement actions in pursuit of the CBD's objectives. The term "full agreed incremental cost" defines the core of the principle.

The definition of this term in the CBD adopts a quite general form and offers a margin for negotiation when it is applied in the context of a specific project. Accordingly, the GEF council provides certain specifications on how to interpret these terms and use the incremental cost principle in practice (GEF 1996). The major points are as follows.

The amount that the GEF transfers to the resource country is derived from the comparison of two scenarios: The first scenario is when conservation spillovers are not addressed in the domestic policy of the resource country. No international financial support is given and the incentives in the resource management are unaltered. This benchmark scenario is named the *baseline*.

The second scenario involves the activities that are incremental to the baseline in order to generate or secure global benefits from biodiversity conservation. Baseline and incremental activities together form the *GEF alternative*. The alternative requires that (1) the incremental activities generate an incremental global benefit compared to the baseline and (2) that the resource country, upon

priorities and eligibility criteria and an indicative list of incremental costs established by the Conference of the Parties."

whose territory the activities are carried out, at least attains the total benefits equivalent to those in the baseline.

The guidelines in the GEF (1996) argue that, for practical reasons, both the domestic and the global benefit need not be quantified in monetary terms. It is sufficient if each benefit is identified. In contrast to this, the *cost* that represents the economic burden of the agreed activities has to be described as completely as possible (*full* cost approach). This means that all significant types of costs have to be identified. For example, the indirect costs, i.e., the costs beyond the (geographical) project boundaries, have to be taken into account. Boundaries should be considered insofar as the cost assessment is still tractable. The same applies to the assessment of environmental and social costs that result from the project activities.

Furthermore, they argue that opportunity costs need not be addressed in the incremental cost framework whenever its scope is chosen in such a way that "the most significant" costs of the alternative course of action and the "the most significant" avoided costs of the baseline are captured.[196]

Regarding the challenges and the practical problems in cost assessment, the GEF (1996) acknowledges that the assessment may not yield clear-cut results. More specifically, there are always some uncertainties and, thus, in some cases the estimated project cost can deviate from the actual cost observed afterwards.

However, since the implementation of a conservation project rests upon the willingness of the resource country to carry out project activities within its territory, policymakers should avoid policies that require the country to bear (incremental) costs for which it is not reimbursed. Both sides, the representatives of the multilateral organization (GEF and Implementing Agency) and of the resource country, have to *agree* upon the cost figures identified. It is claimed, that in this regard, an agreement can be reached more easily, if "controversial economic valuation methods (such as those for monetarizing [domestic] environmental benefits and calculating opportunity costs)" are not applied and the cost assessment relies on "just the identifiable monitorable expenditures" (GEF 1996: para. 36).

Conceptual Issues in Applying the Incremental Cost Principle

Given the specifications and interpretations of the principle, incremental costs are nevertheless difficult to determine in a nonambiguous way. Several aspects are not precisely defined and, thus, are subject to negotiations in a project-specific context. The literature organizes these issues around two major questions. These

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196 Avoided costs accrue when project actions replace the activities of the baseline, more specifically, when they make the latter redundant.

are, first, how to define the baseline scenario and, second, how to treat any incremental domestic benefit (Cervigni and Pearce 1995; Øygard and Bromley 1998).

Considering the *definition of the baseline*, the guidelines in the GEF (1996) define a set of criteria that the baseline scenario has to satisfy:

- It must address "national development goals."
- It must be "technically feasible."
- It must represent an "economically attractive course" that remains "broadly consistent with political and social constraints."
- It must be "environmentally reasonable," i.e., it "does not penalize progressive environmental action."
- It must be "financially realistic."[197]

In addition, the baseline should be established for overall development objectives. To avoid the potential strategic behavior of the resource countries, it has to be ruled out that the baseline is defined for the GEF incremental cost assessment only.

To some extent, these criteria have the character of rationality requirements. They, however, represent only vague constraints on the set of feasible and rational actions. A question that is still to be answered is what activities should be included in the baseline. When referring to the discussion of cost categories and scope of the cost assessment, the boundaries that should underlie the description of the baseline in the spatial and temporal dimension need to be determined. Both the transfer donor and the resource country have to reach an agreement on the activities that are finally to be integrated in the baseline scenario. Although the GEF operational guidelines impose some restrictions upon this decision-making process, they leave a substantial margin for negotiation (Øygard and Bromley 1998).

Another important aspect with respect to the baseline assessment concerns the existence of *policy failures* that are primarily expressed in domestic price distortions in the resource countries. These distortions often create perverse incentives for nonsustainable resource use, which supports biodiversity degradation (Øygard and Bromley 1998). By removing these distortions, sometimes a domestic net benefit can be created while, at the same time, conserving (globally important) biodiversity. If the removal of distortions appears to be rational from a domestic point of view and biodiversity is conserved to a sufficient extent, any international support in terms of financial resources is eventually not required

[197] According to Øygard and Bromley (1998), the baseline "may also include actions that have not yet attained a secured funding."

(Kumari and King 1997). In this vein, it is proposed that GEF funding should not reward distortions in domestic pricing policy (GEF 1996: para. 21).

Otherwise, some question whether a resource country is willing and/or able to remove barriers in domestic pricing policy without external (financial) help. In order to enforce a domestic policy change, who experiences a gain in benefits and who suffers from a relative loss in well-being if the policy change takes place is often of importance. Likewise, resource countries often lack precise information to identify the best policy strategy, especially if it is difficult to assess cost savings in the baseline course that would result from a change in domestic policy (Cervigni and Pearce 1995; Kumari and King 1997).

The other major issue refers to the case where incremental activities not only yield (global) benefits for the rest of the world but also *incremental domestic benefits*. Empirical evidence from official project suggests that such benefits occur in many of the approved projects. Since the incremental cost of biodiversity conservation in a project context describes the amount that the GEF has to transfer to the resource country, the question is what to do if the resource country also experiences an increase in well-being.

A rigorously cost-effective use of the scarce GEF resources requires any incremental domestic benefit to be deducted from the derived incremental cost, i.e., the transfer to the resource country should correspond to the *net incremental cost*. However, the incremental activities, or, synonymously, the GEF-supported project actions, would leave the resource country as well off as without the activities being implemented. In this respect, the resource country may have few incentives to cooperate. On the contrary, a resource country apparently prefers to receive a conditional transfer equal to the *gross incremental cost*, i.e., the incremental cost without deduction; it would thereby appropriate a positive return for conserving biodiversity in excess of the domestically optimal level (Cervigini 1998).

The problem of how to treat potential domestic benefits in the incremental cost assessment is addressed in the GEF (1996: paras. 25–30). Generally, different categories of incremental domestic benefits are identified and specific instructions given for each of them:

- First, the incremental domestic benefit may represent cost savings that relate to an activity or an objective explicitly addressed in the baseline scenario but that are due to the incremental activities in the GEF alternative. In this case, the cost savings should be fully identified and "factored" into the incremental cost analysis.
- Second, it is assumed that incremental activities may yield additional domestic benefits that are not addressed in the baseline. These benefits are assumed not to be of "national priority" because they are considered "uncertain, un-

quantifiable, unimportant or unfinanceable." Since "no further costs would be avoided," these benefits should not enter into the incremental cost analysis.

- Third, if incremental activities increase the supply of those ecosystem services displaying the character of a global public good, the resource country as a member of the international community also receives an additional benefit. However, this benefit should not be assessed and deducted from the incremental cost (GEF 1996: para.30).

Generally, the instructions in the GEF (1996) are somehow ambiguous. On the one hand, it is argued that a transfer to the resource country is "to be for the incremental cost and not for any lesser amount calculated by subtracting either any additional domestic benefit or share of global benefit that the country enjoys" (GEF 1996: para.30). This seems to plead for transfers equal to the size of the gross incremental cost; on the other hand, it is said that the incremental cost approach should guarantee that "scarce funds will be dedicated to global environmental benefits rather than to achieving development and local environmental benefits, for which other sources of funds are appropriate" (GEF 1996: para.6). This latter statement apparently emphasizes the cost-effective use of the resource by the GEF, which requires the resource country to be compensated for the net incremental cost only.

Even when following the instructions in the official documents, it seems to be up to the representatives of the GEF and the resource country to negotiate how additional domestic benefits should be treated. More precisely, both sides have to agree upon how to separate any domestic benefit from the global benefits and what national priority should be assigned to different incremental domestic benefits.

To resume, the incremental cost principle in the context of the CBD-GEF relationship acknowledges the existence of biodiversity spillovers from the developing world. Moreover, it supports the (partial) community-pays principle: the developed countries, as beneficiaries, pay compensation if resource countries in the developing world forgo certain resource uses that are detrimental to globally important biodiversity.

Given the sovereign rights of the resource countries and the need for voluntary cooperation between countries, the incremental cost principle operates as a guideline or institutional rule in international cooperation. More precisely, it serves as the instrument to determine the amount of financial resources to be provided for activities that are implemented in accordance with the CBD's objectives. While the specification and use of the principle offers a margin for negotiation in a project-specific context, the incremental cost framework narrows the range of possible cost estimates and, thereby, the set of possible outcomes of the negotiations. In this respect, the use of the incremental cost principle has an influence upon both the extent of conservation (efficiency) and the distribution of the benefits generated from incremental conservation (equity). The following two sections investigate this interplay in more detail.

Finally, Kumari and King (1997) argue that, in addition to its function as an operative instrument in cost assessment, the incremental cost framework also represents a strategic instrument to guarantee a rational project design.[198]

4.3.2 The Incremental Cost of Biodiversity Conservation from a Theoretical Perspective

The following section reviews the theoretical literature on international cooperation and transfer payments for conservation. The literature typically considers economic problems in the allocation of transfers for a bundle of predefined conservation activities, i.e., these studies do not investigate the decision by donors to select from several international conservation options.[199]

In a noncooperative outcome without transfers to the resource country, it does not internalize the positive spillover generated from their resources. In a cooperative outcome, donors supply transfers to the resource country, which agrees to preserve additional biodiversity resources. Given rational decision making, this cooperative outcome characterizes a Pareto improvement.

Against this background, negotiations between the resource countries as recipients of transfers and developed countries or multilateral institutions as transfer donors determine the exact level of conservation and the transfer amount. Both parameters actually determine to what extent the well-being in the individual countries increases in a cooperative outcome relative to the noncooperative benchmark. The aggregate increase represents the surplus from cooperation. The actual distribution of this surplus is influenced by the allocation of bargaining power between the two sides. The incremental cost principle in this context represents an institutional rule that determines the set of possible bargaining solutions.

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[198] By applying the framework, the project design includes the following steps. As a starting point, an environmental problem of global importance is identified, for instance, the declining population of an endangered species, and the underlying causes of the problem, i.e., the causes of threats to the species, are described and investigated. Based upon this, activities are formulated that remove the underlying threats and, thus, support the preservation of the species (GEF alternative). At the same time, activities are identified that also address the removal of these threats and that are implemented without external support (baseline). Finally, the differences between the two scenarios, i.e., the incremental activities, are identified.

[199] The literature on reserve design studies the issue of selection in detail, although mostly in abstraction from the international dimension (see Section 4.1.1).

4.3.2.1 Conservation and Transfers in a Static Partial Equilibrium Setting

Several theoretical studies conceptualize the allocative and distributional implications of international bargaining on biodiversity conservation and transfers by using a simple partial equilibrium framework (Amelung 1991, 1993; Cervigni and Pearce 1995; Siebert 2005: 206ff.). Within a *two-country setting*, conservation in a resource country, *R*, which hosts valuable biodiversity, is represented by a one-dimensional objective variable that is defined along a continuum. The level of conservation is controlled by the size of the transfer that the rest of the world, *ROW*, provides.

I recapitulate the reasoning in the literature by means of a *simple model*: let q denote the current level of conservation with $0 \le q \le q_0$, where q_0 denotes the initial or maximum level. On the one hand, conservation generates global public goods that contribute to the nonuse values of biodiversity and local private goods as, for example, tourism services or sustainable harvested nontimber forest products.[200] The benefit from conservation for $i = R, ROW$ is described by concave functions $B_i(q)$ with $dB_i(q)/dq > 0$ and $d^2B_i(q)/dq^2 \le 0$; on the other hand, conservation is associated with opportunity costs due to forgone revenues from alternative land use (Amelung 1991). The costs of conservation are $C(q)$ with $dC(q)/dq > 0$ and $d^2C(q)/dq^2 \ge 0$, and accrue to the resource country only.

Based upon assumptions on how to aggregate the domestic and international benefits in order to describe global benefits from conservation, the domestically and globally optimal levels of conservation can be determined. In the *domestic optimum*, q_d^*, marginal cost equals domestic marginal benefit, while in the *global optimum*, q_g^*, marginal cost equals the sum of the domestic and *ROW*'s marginal benefit:[201]

$$(4.1) \quad q_d^* = \left\{ q : \frac{dB_R(q)}{dq} = \frac{dC(q)}{dq} \right\},$$

$$(4.2) \quad q_g^* = \left\{ q : \frac{dB_R(q)}{dq} + \frac{dB_{ROW}(q)}{dq} = \frac{dC(q)}{dq} \right\}.$$

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[200] In addition, conservation may jointly provide local public goods or club goods, such as water purification. For simplicity, these goods are ignored in the model at hand.

[201] It is typically assumed that marginal benefits are positive but continuously decreasing in conservation. In contrast, the marginal cost is constant or continuously increasing in conservation. To determine the global optimum in this regard, Cervigni and Pearce (1993) assign equal weight to the domestic benefit and the benefit of the *ROW*, i.e., a Benthamite welfare function underlies the optimization.

The decision of the sovereign resource country on a conservation level drives the allocation: the country maximizes its net benefits by choosing q_d^*. The donors, *ROW*, influence the incentives of the resource country by providing *transfers*. Cooperation between both parties is then expressed in an agreement on a level of conservation in excess of the domestic optimum and the size of the transfer payment to the resource country.

Any agreement that is to be concluded in this regard has to satisfy the *participation constraint* of both parties, i.e., when that agreement is enforced, neither the resource country nor the transfer donors are worse off relative to the domestic baseline.[202] The two constraints with regard to the (net) benefits finally determine the spectrum of potential bargaining solutions. Amelung (1991, 1993) in this regard makes the strict assumption that both parties necessarily agree upon a level of conservation that is equal to the global optimum, and that only the size of the transfer and, thus, the distribution of the cooperation surplus is subject to negotiations. In contrast, Cervigni and Pearce (1995) describe in general terms that the parties reach an agreement on a certain level of conservation that is above the domestic optimum but that may be equal to or below the global optimum.

Considering their relationship to the policy of the incremental cost principle, these theoretical representations in the literature assume that the domestically optimal level of conservation represents the baseline scenario (without transfer policy). Given perfect information regarding the benefit and cost of conservation and the absence of domestic policy failure, the proper definition of the baseline is not subject to discussions.

Furthermore, taking potential incremental domestic benefits into account is implicitly included in the possible bargaining solution. In other words, the studies typically do not predict the outcome of bargaining precisely but only describe the range of possible outcomes. In this respect, a transfer agreement that merely satisfies the resource country's minimum willingness to accept a given level of extra conservation corresponds to the net incremental cost approach. In contrast, an agreed upon transfer amount in accordance with the gross incremental cost approach is equal to the total expenditures for the agreed upon level of extra conservation.

[202] For any level of conservation between the domestic optimum and the global optimum, the donors are only willing to make a transfer for conservation if the value of the additional benefit they receive exceeds the value of the transfer. Likewise, the resource country only agrees to extra conservation efforts if the transfer exceeds the resulting net incremental cost, i.e., the incremental expenditures to attain the extra conservation minus the incremental domestic benefits that are generated by this (Cervigni and Pearce 1995).

Since the *ROW*'s maximum willingness to pay is defined by the incremental benefit it receives, it is unclear whether, in a project-specific context, this maximum willingness to pay is equal to or greater than the incremental expenditures, which, according to the incremental cost principle, define the maximum transfer amount. The following equations demonstrate that, under certain conditions, the difference between the two variables is indeed positive, i.e., the maximum willingness to pay for a certain level of extra conservation can be greater than a transfer that is equal to gross incremental cost.

For this purpose, I specify the utility functions $B_i(q)$ for $i = R, ROW$ in a way that marginal benefits are represented by linear functions:

$$B_R = aq - \frac{b}{2}q^2, \quad B_{ROW} = sq - \frac{v}{2}q^2. \tag{4.3}$$

For a constant marginal cost, c, the domestic optimum is derived as

$$q_d^* = \frac{a-c}{b}. \tag{4.4}$$

The associated benefits are

$$B_R(q_d^*) = \frac{a^2 - c^2}{2b}, \quad B_{ROW}(q_d^*) = \frac{(a-c)(2bs - (a-c)v)}{2b^2}. \tag{4.5}$$

Suppose, then, that *ROW* and the resource country agree to implement the globally optimal level of conservation (see equation (4.2)). Given the functional forms, this is

$$q_g^* = \frac{a-c+s}{b+v}. \tag{4.6}$$

The resulting benefits for *ROW* and for the resource country are

$$B_R(q_g^*) = \frac{(a-c+s)(b(c-s) + a(b+2v))}{2(b+v)^2}, \tag{4.7}$$

$$B_{ROW}(q_g^*) = \frac{(a-c+s)(2bs + (a-c-s)v)}{2(b+v)^2}.$$

Using these equilibrium levels for conservation, the incremental cost is defined as

$$(4.8) \quad c\left(q_g^* - q_d^*\right) = \frac{c(bs-(a-c)v)}{b(b+v)}.$$

Using the derived benefits for both equilibria, the incremental domestic benefit is

$$(4.9) \quad B_R\left(q_g^*\right) - B_R\left(q_d^*\right) = \frac{1}{2}\left(\frac{(a-c+s)(b(c-s)+a(b+2v))}{(b+v)^2} - \left(\frac{a^2-c^2}{b}\right)\right).$$

In the same way, the incremental benefit for *ROW* is:

$$(4.10) \quad B_{ROW}\left(q_g^*\right) - B_{ROW}\left(q_d^*\right) = \frac{(2b+v)(bs-(a-c)v)^2}{2b^2(b+v)^2}.$$

A transfer in accordance with the *net incremental cost* approach represents the difference between the incremental cost and the domestic incremental benefit:

$$(4.11) \quad c\left(q_g^* - q_d^*\right) - \left(B_R\left(q_g^*\right) - B_R\left(q_d^*\right)\right) = \frac{(bs-(a-c)v)^2}{2b(b+v)^2}.$$

This transfer amount represents the resource country's minimum willingness to accept the implementation of a globally optimal level of conservation, q_g^*, within its territory.

A transfer according to the *gross incremental cost* approach is equal to the incremental cost (see equation (4.8)). *ROW's* maximum willingness to pay for the implementation of q_g^* is equal to the incremental benefit it receives.

To verify whether there is a positive margin between the maximum willingness to pay and the gross incremental cost, I calculate the difference between the two terms

$$(4.12) \quad \left(B_{ROW}\left(q_g^*\right) - B_{ROW}\left(q_d^*\right)\right) - c\left(q_g^* - q_d^*\right) =$$

$$\frac{(bs-(a-c)v)((bs-(a-c)v)(2b+v)-2bc(b+v))}{2b^2(b+v)^2}.$$

The margin in equation (4.12) is zero whenever the numerator is zero. In order to confirm that a positive margin can exist for a set of reasonable parameter values, it has to be shown that the numerator is positive. For this purpose, let us initially assume that all parameters are positive.

Furthermore, from $q_d^* > 0$, it follows that $a > c$. It can also be shown that a positive incremental cost, as well as a positive domestic incremental benefit, requires that $0 < v < bs/(a-c)$. I give the numerator as a function of v:

(4.13) $f(v) = (bs-(a-c)v)((bs-(a-c)v)(2b+v)-2bc(b+v))$.

The three zeros of this function are

(4.14) $$v_{01} = \frac{bs}{a-c} > 0,$$

$$v_{02} = \frac{b}{2(a-c)}\left[s-2a+\sqrt{4a^2-8ac+8c^2+4as-8cs+s^2}\right],$$

$$v_{03} = \frac{b}{2(a-c)}\left[s-2a-\sqrt{4a^2-8ac+8c^2+4as-8cs+s^2}\right].$$

The first zero is positive but it also represents the upper limit of the range of v, for which the incremental cost is positive. For convenience, let us assume that $2a = s$, i.e., the marginal benefit *ROW* derived from the first unit of conservation is twice as high as the benefit for the resource country. It then turns out that the other two zeros lie symmetrically around the origin with $v_{02} = -v_{03}$.[203]

Since the slope at the zero v_{01} is positive,

(4.15) $$\frac{df(v_{01})}{dv} = 2b^2c(a-c+s) > 0,$$

there must be a positive range of v that is connected with a positive margin, i.e. $f(v) > 0$, whenever $\{a,b,c,s\} | 0 < v_{02} < v_{01}, df(v_{02})/dv \neq 0$.

By using the symmetry assumption, it be can illustrated that the inequalities hold. The condition that the slope at v_{02} is different from zero may constitute the normal case but is difficult to derive in formal terms.

For this reason, I perform a simple numerical simulation (Table 19). I assume that $a := 5, b := 0.7, s := 10, v := 1$. Different values for c are considered. The last row in the table indicates the incremental cost that describes the transfer amount according to the incremental cost principle. The figures illustrate that the incremental cost is smaller than the incremental benefit for *ROW*, which is depicted in the row above.

To resume, the findings support the following conclusions:

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[203] Given that $2a = s$, it can be shown that the root in equation (4.14) is positive.

Table 19:
Benefits and Costs of Incremental Conservation: A Numerical Simulation with a Linear Model and Varying Marginal Cost

Marginal cost (c)	1.00	1.50	2.00	2.50	3.00	3.50	4.00	4.50
Domestic optimum (DO)	5.71	5.00	4.29	3.57	2.86	2.14	1.43	0.71
Global optimum (GO)	8.24	7.94	7.65	7.35	7.06	6.76	6.47	6.18
Incremental conservation	2.52	2.94	3.36	3.78	4.20	4.62	5.04	5.46
Utility R in DO	17.14	16.25	15.00	13.39	11.43	9.11	6.43	3.39
Utility R in GO	17.44	17.63	17.77	17.84	17.85	17.81	17.70	17.53
Incremental benefit R	*0.30*	*1.38*	*2.77*	*4.45*	*6.43*	*8.70*	*11.27*	*14.14*
Utility ROW in DO	40.82	37.50	33.67	29.34	24.49	19.13	13.27	6.89
Utility ROW in GO	48.44	47.88	47.23	46.50	45.67	44.77	43.77	42.69
Incremental benefit ROW	*7.63*	*10.38*	*13.56*	*17.16*	*21.18*	*25.63*	*30.51*	*35.80*
Incremental cost	2.52	4.41	6.72	9.45	12.61	16.18	20.17	24.58

- Under the conditions described by the numerical simulation, the donor (described by *ROW*) receives a positive surplus from cooperation that is not the subject of the negotiations. This is because the incremental cost principle does not enable transfers in excess of the incremental costs. This implies that for any outcome where the donor's maximum willingness to pay exceeds the incremental cost, it is ruled out that the maximum willingness to pay is exhausted beyond the incremental costs.
- In other words, under certain conditions, the incremental cost principle reduces the set of incentive-compatible and, thus, feasible allocations by ruling out potential outcomes that would provide the resource country with a share of the cooperation surplus that is even larger than in the case of a gross incremental cost approach.

4.3.2.2 Management of a Natural Resource Stock and Transfer Payments

The partial equilibrium modeling considers one-time transfers in a one-period setting. A multiperiod setting can be represented by a sequence of static partial equilibria with constant parameters. A second class of studies that rely upon *resource economic foundations* performs a more explicit modeling of the paths of both conservation and transfer payments over time.

In these studies, conservation is no longer perceived as a one-dimensional variable, e.g., the share of land kept under protection (Cervigni and Pearce 1995), but rather as a natural resource, whose stock, $Q(t)$, is subject to changes over time (Coram 2003; Stähler 1996; van Soest and Lensink 2000). Changes occur due to *resource extraction*, $E(t)$, or resource regeneration, $R(t)$, whereas regeneration depends upon the size of the stock, i.e., $R(Q(t))$. $Q(t)$ is nonnegative and $Q_{\max} > 0$ denotes the maximum size of the stock.

The studies assume that the benefits the resource country receives from extraction are perceived as private domestic goods from nonconservation. These goods describe the opportunity costs of conservation. When assuming that the utility function, $U_R(Q(t))$, of the resource country is well-behaved and that this country manages the resource stock in an optimal way, the size of the stock (conservation) converges to a long-term equilibrium (*steady state*). The formal representation of the solution follows the simple model of renewable natural resource management, which is omitted here for reasons of space.

In modification of this standard model, the preservation of the stock may generate *public goods* for the resource country (Coram 2003; van Soest and Lensink 2000). Not surprisingly, with domestic private and public goods from conservation, the resource stock in a steady state is comparatively larger than in a setting without.

When applying the model with resource stock preservation to the incremental cost approach, let us perceive the equilibrium stock as the baseline. As in the static setting, for complete information and efficient domestic policy, the literature does not consider the determination of the baseline controversial.

Turning to the *international level*, the rest of the world, referred to as *ROW*, only receives global public goods from conservation that are approximated by the size of the resource stock, $Q(t)$. In order to influence the resource country's extraction path in a favorable way, *ROW* provides a flow of transfers, $T(t)$. Conservation generates a positive but decreasing marginal benefit for *ROW*. Its utility, $U_{ROW}(Q(t),T(t))$, decreases at a progressive rate with the size of the transfer. This decreasing marginal benefit approximates *ROW*'s use of financial resources for alternative consumption.

ROW and the resource country negotiate the size of the transfer flow and the stock preserved. The resulting outcome at the same time determines the accounting of incremental domestic benefits from extra conservation. The studies typically assume that *ROW* is able to act as the *Stackelberg leader* (Coram 2003; van Soest and Lensink 2000). Only Stähler (1996) considers a subscenario where the resource country has strong bargaining power and is able to exhaust *ROW*'s willingness to pay for conservation.

When *ROW* is a leader, it maximizes its own discounted utility subject to the transfer-induced actions of the resource country, R, which *ROW* anticipates correctly:

$$(4.16)\quad \max_{T} \int_{0}^{\infty} e^{-rt}\, U_{ROW}(Q(t),T(t))\, dt$$

$$s.t.\ \ V \equiv \max_E \int_0^\infty e^{-rt}\, U_R\big(E(t),T(t)\big)\, dt \ s.t.\ \ \dot{Q}(t) = -E(t) + R\big(Q(t)\big) >$$

$$V_0 \equiv \max_E \int_0^\infty e^{-rt}\, U_R\big(E(t)\big)\, dt \ s.t.\ \ \dot{Q}(t) = -E(t) + R\big(Q(t)\big).$$

The constraint $V > V_0$ describes that the transfer size has to be chosen in such a way that the participation constraint of the resource country is satisfied.

The generic transfer function $T(t)$ can be specified in order to represent *different transfer regimes*. Either *ROW* pays compensation per unit of preserved resource stock in a given period, or it penalizes resource extraction by allowing the transfer size to depend upon the extent of forgone resource extraction (Coram 2003; van Soest and Lensink 2000). Following the example in van Soest and Lensink (2000), both elements can be combined in a transfer menu:

$$(4.17)\quad T(t) = aQ(t) - bE(t) - d\big(E(t)\big)^2 .$$

Compensation per unit can be defined as a constant price or an increasing price for a decreasing resource stock. The latter scheme reflects a positive but decreasing marginal benefit of conservation for *ROW* as the transfer donor (Stähler 1996).

Generally, the studies assume bargaining power in favor of *ROW*. They describe the steady state resource stock without transfers, when transfers are paid and when various transfer schemes are implemented. The following results are derived from the studies: (1) (Linear) transfers based upon the size of the preserved stock increase the size of the stock in equilibrium. (2) Extraction-based transfers increase the equilibrium stock only if either domestic public goods from conservation exist or if the compensation per unit of forgone extraction is very large, i.e., it is greater than the marginal domestic benefit of extraction (Coram 2003).[204] (3) Comparing extraction-based and stock-based transfers, one scheme cannot be really preferred over the other. To conclude, the studies demonstrate that the effectiveness of transfers payments, defined as additional conservation per dollar, depends upon the stock size and the extent of the extraction in the baseline equilibrium, as well as upon the functional form of the transfer scheme (Coram 2003). Furthermore, under certain conditions, a combination of extraction-based and stock-based transfer schemes is optimal from the point of view of the donor (van Soest and Lensink 2000).

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[204] Coram (2003) shows that if domestic public goods from conservation exist, a "buy-out" of the resource by international donors, i.e., purchases of parts of the resource stock to preserve them from exploitation, largely dominates domestic investments in conservation and is, therefore, rather ineffective.

When bargaining power is in favor of the resource country, there is a chance of a "failure" in transfer policy. It turns out that for elastic stock-based transfer schemes, i.e., the transfer increases with a decreasing resource stock, it may create incentives in such a way that it is optimal for the resource country to preserve a lower stock than in a baseline with no transfers. The choice of an appropriate transfer scheme can prevent such adverse impacts (Stähler 1996).

When investigating the treatment of *incremental domestic benefits* from conservation, I conclude that the modeling in the studies is sufficiently flexible to rule out such benefits by simply blinding out both domestic private and public goods from conservation. This has been done for the case of monopolistic behavior on the part of the resource country (Stähler 1996). In this case, transfers serve to compensate for the incremental cost of providing global domestic benefits, which are approximated by the size of the resource stock.

In order to condense the theoretical findings on the allocation of bargaining power and the existence of domestic benefits for the practical application of the incremental cost principle, I reach the following conclusions:

- The combination of domestic benefits from conservation and bargaining power for the donor (Stackelberg leadership) implies that a net incremental cost approach is pursued, i.e., the financial resources for transfers are used in a cost-effective way. In contrast, the absence of domestic benefits from conservation, together with bargaining power for the resource country, can lead to a counterproductive impact of transfers upon the obtained level of conservation.
- The findings on the Stackelberg behavior of the resource country may be interpreted as a prediction of the increasing bargaining power of certain resource countries if global biodiversity continues to decline. In this context, the effectiveness of transferred funds is decreasing, regardless of whether a net or a gross incremental cost approach is applied.
- The provisions on the determination of the transfer amount given by the incremental cost principle only relate to the costs of incremental conservation. In this regard, the provisions on the stock-based transfers, as described in the resource economic model, are eventually not supported by the provisions. In other words, the principle seems to restrict the set of transfer schemes available to the extraction-based ones.

4.3.2.3 Transfer Policy in a Two-country Two-commodity Setting

A shortcoming of both the static and dynamic partial equilibrium models is that they more or less only investigate extreme point allocations of bargaining power. In this regard, the model by Cervigni (1998) allows for cases of an interior allocation of bargaining power. A strength of this model is that it explicitly describes the substitution between biodiversity conservation and alternative nonenvironmental consumption. The previous partial equilibrium approaches assume a zero price elasticity of the demand for conservation in this respect.[205]

The model again relies upon the static partial equilibrium approach but introduces an additional, exclusive nonconservation good that can be purchased by both the resource country and *ROW* at identical and constant prices. Furthermore, each of the two countries is endowed with a given national income. The resource country allocates its income to *alternative consumption* and the appropriation of *private conservation goods*, which are linked to the biological resources produced in natural areas conserved for this purpose. The appropriation of these commodities comes at a constant price per unit of conserved area. Conserved areas initially do not yield domestic public goods but only *global public goods*, which *ROW* receives. In the baseline allocation without transfers, *ROW* spends its entire income on the alternative consumption good.

Given this setting, Cervigni derives the domestic and the *global optimum* for conservation. In order to determine the global optimum, he assumes a global welfare function of the Benthamite type. Because of well-behaved utility and cost functions, both the domestic optimum and the global optimum are unique.

To describe the international transfer policy, he introduces a *transfer function* in the constraint that describes the budget of disposable resources for each of the two countries. The transfer function in the constraint of the resource country shows a positive sign, while a negative sign is indicated for the transfer function in *ROW*'s constraint. The function is constructed in such a way that each level of transfer represents the corresponding level of conservation agreed. Since a negotiation outcome requires that the participation constraint of the resource country is met, the transfer function also embodies the property that the resource country at least attains the same utility as in the domestic optimum for each combination of transfer and conservation.[206]

To describe the extent to which any *incremental domestic benefit* is taken into account and deducted from the gross incremental cost in order to determine the size of the transfer, the author introduces the parameter $\gamma \in [0,1]$ into the transfer function. It is shown that the level of conservation that both parties agree upon depends simultaneously upon the agreement on how to treat the incremental domestic benefits from the implemented activities, i.e., the value of γ.

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205 The previously described modeling assumes that, for a given level of incremental conservation, the marginal benefit from conservation for both countries does not change when the size of the transfer varies, i.e., when the distribution of the cooperation surplus changes.

206 Methodologically, this is achieved through the use of a dual approach to the resource country's welfare maximizing problem (Cervigni 1998).

In this respect, the treatment of these benefits does not only determine how the surplus from cooperation is distributed but also the size of the surplus.

Two different transfer schemes, or settings, are considered. First, the model assumes that the resource country is able to choose a value for γ that maximizes its utility.[207] Simultaneously, *ROW* only decides upon the transfer size in order to determine the level of conservation. *ROW*'s willingness to pay for transfers depends upon how much additional conservation it obtains for a given amount of funding. If a decreasing share of incremental domestic benefits is taken into account, i.e., $\gamma \to 0$, the unit cost of incremental conservation from the point of view of *ROW* increases. Since ROW can spend its income on alternative consumption, conservation is, ceteris paribus, substituted for by consumption and less funding for transfers is provided. Second, the model assumes that the resource country cannot control γ and is willing to accept a transfer with $\gamma = 1$, where incremental domestic benefits are completely taken into account. It is shown that under these conditions, the transfer can establish the theoretical global optimum.

Returning to the first setting, where the resource country can control γ, the agreed upon outcome corresponds to a relatively low level of conservation as compared to the setting in which benefits are almost completely accounted for, $\gamma \to 1$. This outcome is associated with a lower incremental benefit for both *ROW* and the resource country. Regarding the motivation of the resource country in this respect, it is crucial whether a large share of a smaller "cake," i.e., a surplus from incremental conservation, is greater in absolute terms than a small share of larger "cake." Against this background, the optimal treatment of benefits from the point of view of the resource country is when the outcome is greater than zero.

Cervigni demonstrates that for $\gamma \to 0$, conservation approaches an equilibrium level where the marginal cost of conservation is equal to the marginal benefit of *ROW* (Cervigni and Pearce 1995). Otherwise, for $\gamma \to 1$, i.e., taking the incremental domestic benefit almost completely into account, *ROW* merely maximizes its utility subject to the participation constraint of the resource country. In other words, for $\gamma \to 1$ the transfer policy puts strong emphasis on cost-effectiveness, and the level of total conservation approaches the theoretical global optimum. The question is whether the resource country is willing to accept such an allocation or whether it has some veto power and can influence γ.

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207 Two basic schemes are distinguished (Cervigni 1998). Either *ROW* asks for the level of conservation that is to be implemented and the host country claims the conditional transfer, i.e., it behaves as a quantity taker, or *ROW* offers a conditional transfer menu and the host country chooses its optimal level of conservation (transfer-taking behavior).

In the treatment of the incremental domestic benefit, the modeling implies the following:

- The decision on how to take incremental domestic benefits into account, i.e., whether to apply the gross or net incremental cost approach, at the same time influences the level of conservation that the two parties will agree upon. The transfer payments will implement the globally optimal level of conservation only when compensating for the net incremental cost, $\gamma = 1$. In other words, an agreement on the global optimum that does not take certain incremental domestic benefits into account in the transfer size cannot be established as an outcome; this being the case, it is rational for *ROW* to reduce its transfer payments. Note, however, that this conclusion relies upon a relatively strict definition of the global optimum: because of the Benthamite welfare function used in the model in order to determine the global optimum, it is implied that one utility unit of the resource country can be substituted for perfectly by one utility unit of *ROW*. If this assumption is relaxed, the net incremental cost approach , $\gamma = 1$, need not correspond to achieving the global optimum.
- Since $\gamma = 1$ typically is not the utility maximizing choice from the view of the resource country, or more explicitly, since the resource country gains little from cooperation, it may have few incentives to enter into an agreement on extra conservation and transfers, i.e., an agreement with $\gamma = 1$ may hardly occur in practice.

From this it is concluded that neither of the extreme values for γ may prevail in practice. The implication for the transfer policy in practice is that neither the net incremental cost nor the gross incremental cost approach in its pure definition is widely applied. Moreover, to maximize their own well-being, it is desirable for both sides to allow the opponent to appropriate some parts of the surplus.

Regarding the second issue of how to determine the *baseline*, Cervigni (1998) defines an extension to his model in order to consider domestic price distortions: he assumes that there are also *domestic public goods* that are supplied jointly by conservation. Furthermore, the sustainable use industry responsible for the production of the two types of goods considered is unable to restrict access to the public goods and, therefore, generates additional revenues from conservation. Consequently, the industry operates with a private output price that is below the socially optimal price. If public policy intervention does not provide for internalization, the domestic equilibrium (baseline) does not represent a domestically optimal allocation.[208] In an example with specific functional forms, the author

[208] In addition to noninternalized benefits in the sustainable use sector, price distortions in the nonconservation sector are assumed, for instance, due to a subsidization.

demonstrates that the existence of price distortions leads to the less effective use of financial resources for transfers but does not necessarily impede an agreement on incremental conservation and transfer payments.

Finally, while the modeling indeed provides fresh insights, it may invite criticism of the description of the trade-off between conservation and land development: in the model, a "sustainable use industry" generates both domestic private benefits and international external benefits. The size of that industry and, thereby, the land in sustainable use is not fixed, but, rather, reacts elastically to the demand for conservation. This suggests that, aside from the land occupied by the sustainable use industry, there are further areas that are either converted and used in the nonenvironment industry or completely unused and, thereby, preserved without human interference. However, the modeling neglects these areas and, therefore, neither considers a trade-off between conservation and converting uses of natural areas, which leads to responses in the supply of the alternative consumption good, nor accounts for the generation of international external benefits in the unmanaged areas.

4.3.3 Applying the Incremental Cost Principle: Incremental Cost Analyses

Given the previous conceptual findings on the incremental cost principle, this section studies its empirical side. It uses the sample of protected area projects that are supported in the *GEF's biodiversity focal area* to describe and analyze figures from the applied incremental cost analyses.

The focus falls upon the shares of the different financiers in the projects. Against the background of the discussion on incremental domestic benefits presented in the previous sections, I analyze the proportion of GEF funding relative to funding from domestic sources. More specifically, I am interested in the scope of the resource countries' participation in the financing of the incremental activities. Based upon the empirical findings, I (1) derive general implications regarding the size of the domestic benefits in the GEF projects and (2) discuss the distribution of the cooperation surplus. For the empirical analysis, I draw upon the sample of *295 protected area projects* identified in Section 4.2.3.

Accordingly, the price for nonconservation goods is below the price level that would reflect the actual social scarcity of these goods. Cervigni (1998) notes that although the *ROW* is actually worse off relative to the situation without price distortion, the host country may attain a higher utility level depending upon the preferences for alternative production and conservation.

4.3.3.1 The Practical Use of Incremental Cost Analyses

In order to investigate the practical use of the incremental cost principle, I use the official documents for the identified GEF projects. According to the current GEF guidelines on project application, the incremental cost analysis is required when requesting GEF funding. For some of the 295 projects, the documents either do not contain incremental cost assessments or the presented figures are incomplete or inconsistent. Therefore, I only include 219 projects in my working sample.

While the incremental cost analysis can be perceived as an instrument for *project selection* in the sense that among proposed actions that yield the same global benefit, the one with the lower incremental cost shall be preferred, the incremental cost criterion has been used previously as a heuristic justification and not as a rigorous tool for decision analysis. Moreover, it should be pointed out that although the concept of the incremental cost is important for project selection, other criteria, such as conservation priorities, national goals, equity consideration, environmental, and social acceptability, also matter (Fairman 1996; GEF 1996).

Regarding the project selection, the project documents do not provide evidence that the figures that are derived in the incremental cost analyses are compared to a cost-benefit benchmark or that certain project proposals have been rejected because of an insufficient social return. In this regard, the official documents for some projects are supplemented by official memorandums, which sometimes contain recommendations for a revision of the submitted project proposals.

As described in Section 4.3.1, to conduct an incremental cost analysis, the proposed activities are divided into the baseline course of action and the GEF alternative. The baseline contains activities that are enforced irrespective of GEF funding. The GEF alternative comprises the baseline (and, thereby, achieves at least the same domestic benefit) plus the extra activities to realize the global benefit.

In accordance with the specification in the GEF (1996), the benefit on a domestic and global level is not typically described in monetary terms. More precisely, there is neither information on the values of the output generated in the baseline and the alternative nor on documented evidence that the evaluation methods are applied in order to determine such values. I can only consider the planned *project outputs*, i.e., the practical conservation goals, for example, securing a viable population of an endangered species, as observable proxies for the domestic and global benefit.

Biodiversity conservation represents a cross-cutting issue. Therefore, policies in different areas that pursue other objectives, such as agricultural markets, housing, or poverty alleviation, may have a reinforcing impact upon conservation. Regarding the definition of the baseline, the project documents do not indicate how the baseline is actually separated from such policies. It is unclear whether the bundling of activities in projects follows an approach or a guideline that is unique across the projects.

In order to describe the costs and financing in a GEF project, the project documents introduce several technical terms, in addition to those of the baseline, the GEF alternative, and the incremental cost. These are, namely the increment, the sustainable development baseline, and the total project cost.

Since the baseline and the GEF alternative are both described as expenditures, the *increment* represents the difference between the two courses of action in monetary units. The increment, thus, is a synonym for incremental expenditures. While the incremental cost is always treated as the amount to be financed by the GEF, i.e., while it represents the size of the GEF grant, increment and incremental cost diverge because of a positive incremental domestic benefit.

In this context, Chomitz and Kumari (1996) argue that "for many projects, ...[incremental] domestic benefits either do not exist or cannot be quantified with sufficient rigor to support ... a convincing cost-benefit analysis." Accordingly, the documents only indicate a positive incremental domestic benefit in monetary terms for a few projects.

In the official document, a large relative size of this benefit is indicated for the Local Empowerment and Environmental Management Program project in Nigeria (58 percent of the Increment) and the Komodo National Park Collaborative Management project in Indonesia (57 percent). In contrast, for the Coastal and Biodiversity Management project in Guinea-Bissau (36 percent) and the Lalkisale Biodiversity Conservation Support Project in Tanzania (33 percent), the relative size is comparatively smaller. In all of these examples, the benefit estimate is introduced into the incremental cost assessment and is, thereby, deducted completely from the increment when determining the incremental cost. However, given the problems in assessing the benefits of a project with sufficient accuracy, the documents do not discuss to what extent the estimates derived represent the true benefits.

Most project documents do not provide an assessment of the incremental domestic benefit in monetary terms. Nevertheless, differences between the increment and the incremental cost can serve as evidence of the existence of this benefit, especially if funding from domestic sources fills the gap between the baseline and the GEF alternative.

If the baseline course of action only is considered, it is conceivable that these activities also generate some spillovers and ecosystem services of global importance. In contrast to taking potential incremental domestic benefits of the amount of the increment into account, there is no evidence that the incremental cost analyses in the project document take potential cross-border spillovers generated in the baseline into consideration. This is consistent with the instruction in GEF (1996) that states that transfers should not be supplied for biodiversity conservation that is financed by domestic sources. In order to change the incentives for conservation, the GEF mechanism provides funding for marginal conservation but not for the stock of natural capital, which is protected regardless.

Nevertheless, international donors participate in the cost-sharing of baseline activities outside the GEF framework: for some projects, a *sustainable development baseline* is defined, which represents the domestic baseline leveraged by international cofinancing provided irrespective of the GEF support. This international cofinancing includes unilateral assistence, as well as funding by other multilateral institutions.

Two alternative motivations support the international funding of baseline activities for conservation. First, these activities may contribute to economic development and poverty alleviation and, therefore, are supported by international donors for the same reasons that apply to development assistance (Jayaraman and Kanbur 1999). Furthermore, baseline activities may generate international spillovers that accrue exclusively to individual countries in the developed world (see Section 4.1.3).

Finally, the incremental cost analysis is an assessment separate from the assessment of the *total project cost*, which describes the actual or, at least, planned expenditures that have to be financed from domestic and international funding sources. Consequently, the figures used in the incremental cost analysis are not always consistent with the components of the total project cost. In fact, it is shown that the relationship between the latter and the figures in the incremental cost analysis varies from project to project: in some cases, the total project cost more or less equals the estimated incremental cost, while in other cases, they are close to the cost of the GEF alternative.[209] Nevertheless, the project documents show that the size of the GEF grant in the final funding of the projects is always consistent with the derived incremental cost in the corresponding analysis. In the following, I omit figures on the total project cost and continue with the figures in the incremental cost assessment only.

4.3.3.2 Incremental Cost Assessments: An Empirical Analysis

Considering the set of the *219 projects* with the complete incremental cost assessment provided in the official project documents, their aggregate cost in the

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[209] More precisely, empirical evidence in the project documents shows that the total project cost coincides with the incremental cost for 93 projects. For 100 projects, the total project cost equals the costs of the GEF alternative.

GEF alternative amounts to *$7.72 billion*. This includes the aggregate cost of the baseline activities, which is *$5.85 billion*. Consequently, the aggregate incremental expenditures for the sample considered amount to *$1.87 billion*. Regarding the individual projects, the baseline includes, on average, 52 percent of the cost of the GEF alternative; the remaining 48 percent represent the incremental expenditures, i.e., the increment.[210]

Due to incomplete data in certain documents, I could retrace the complete finance of the baseline with respect to the various funding sources for only 116 projects. For 197 projects, I could reproduce the funding of the increment. I categorized the different financiers into multilateral donors, unilateral donors, domestic public sector institutions and financiers in the private sector (see Section 4.2.3.3). The latter comprises local communities and stakeholders within the project sites and also domestic or multinational firms, foundations, and national and international NGOs.

Because of incomplete data, I omit figures on the coverage of aggregate costs for the baseline and GEF alternative; I only calculate the average shares for each financier group. Particularly, the figures on average financing shares in the increment may provide some information on how incremental domestic benefits are treated. The results on the average shares are presented in Figure 21.

For the *baseline*, the major financiers are the domestic governmental institutions (on average 45 percent). The support for conservation activities from unilateral and multilateral donors other than the GEF is, on average, 21 percent and 14 percent, or 35 percent in total, and consequently below that from domestic contributions. The private sector, on average, provides a substantial share of 18 percent. Again, it is unclear as to whether this private funding originates primarily from domestic or international sources.

Regarding the average financing of a project *increment*, it turns out that the GEF covers more than two thirds of the incremental expenditures (68 percent). Domestic public sector institutions display the second largest share (11 percent), followed by the private sector (9 percent).[211] Local communities in the project areas contribute 2 percent. Other multilateral and official unilateral funding amounts to 4 percent and 6 percent.

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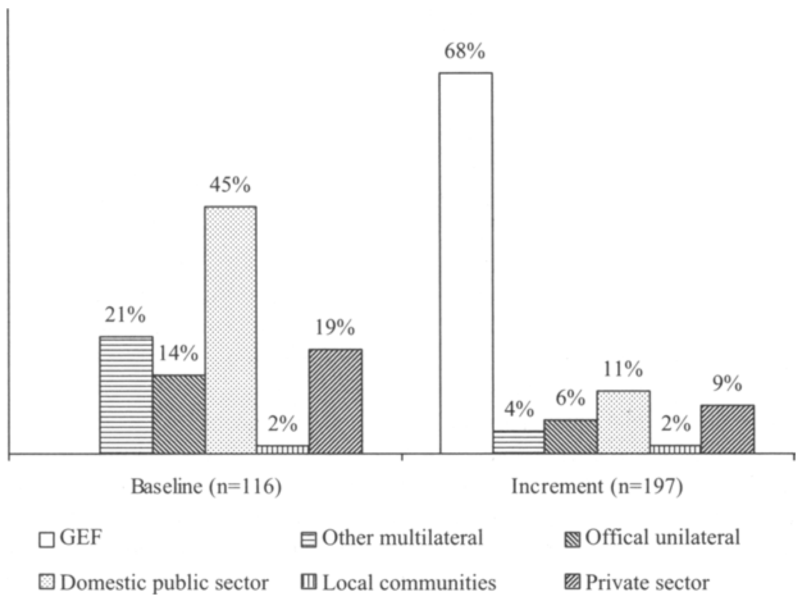
210 By definition, the alternative only includes the baseline if *complementary activities* are added without changing the baseline course of action. In contrast to this, *substitution activities* lead to the reconstruction of the baseline by substituting for the proposed course of action and explicitly taking into account activities that address global environmental externalities. However, since the described activities are only broken down into these categories for some projects, I do not provide a deeper analysis of these terms.

211 With the information provided in the project documents, a distinction according to these categories is hardly feasible.

Figure 21:
Average Financing of the Baseline and GEF Alternative by Financiers



Source: Selected GEF project documents, own calculations.

Domestic Increment Financing and Incremental Domestic Benefits

Given the definitions of the GEF policies, the large share of the GEF funding suggests that the enforced incremental activities generate global spillovers, i.e., the activities contribute to the conservation of biodiversity of global importance. Nevertheless, the figures above indicate that both international and domestic sources finance these activities. Since resource countries are sovereign in their decision making, their participation in the sharing of the project costs serves as evidence that, on average, an incremental domestic benefit indeed exists and that these benefits are apparently charged to some extent in the incremental cost analysis.

Considering the size of the funding of incremental activities from domestic sources relative to that from international sources, the *participation of domestic donors* is reflected in the share of the domestic governmental institutions and that of local communities. In addition, the share of the private sector as far as it represents domestic actors may be taken into account. Furthermore, funding from multilateral donors, such as the World Bank or regional development banks, may constitute loan payments to finance incremental domestic benefits.[212] Consequently, the funding from these other multilateral donors partly reflects the incremental domestic benefit. Nevertheless, the information presented in the project documents is not sufficient to enable a classification of the international payments with respect to grants and loans. Calculating the domestic contributions, i.e., the shares of domestic financiers in the different classifications, I derive a range of 13 to 24 percent for the relative size of incremental expenditures as the average value of domestic contributions.[213]

These contributions match the actual *incremental domestic benefit* a resource country receives from an implemented project only to the extent that it is deducted from the incremental expenditures. Reliable conclusions regarding the *extent of deduction* could, in turn, provide implications for the key issues of the cost-effective use of GEF resources, incentive compatibility, and equity in the GEF project contracts. To answer the empirical question on deduction, the incremental benefit needs to be clearly identified and quantified in monetary terms. However, it is recognized that the incremental domestic benefit of a GEF project can be "uncertain, unquantifiable, unimportant or unfinanceable" (GEF 1996). Furthermore, even if valuation methods are available to estimate benefits in the form of nonmarket values to the resource countries, these methods are not usually applied.[214] Against this background, I cannot describe the actual extent of deduction in a precise way. However, in order to provide at least some insights, I study the share of the GEF in the financing of the increment: for each project with *complete financing* of the increment by the GEF, there is surely no deduction of any positive incremental domestic benefit. A share of less than 100 percent may serve as evidence of deduction under certain circumstances. I discuss the deduction of the incremental domestic benefit below.

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212 This approach is explicitly suggested in the Local Empowerment and Environmental Management Program project in Nigeria. In contrast to this, Chomitz and Kumari (1996) doubt that incremental domestic benefits can generally be determined with reasonable accuracy in order to support a "market-rate" loan.

213 Note that any conclusion regarding the relative size of incremental domestic benefit from financing shares depends upon the consistent design of the project. Suppose that for political reasons, it may be agreed upon to include a bundle of activities in the increment that do not generate any additional global benefit and that are therefore completely financed by the host country. If such activities were not otherwise formally included in the GEF project but stood on their own, the allocation would not change, but the indicated share of the domestic actors in the project financing would be lower. Based upon the information provided in the official project documents, I do not find evidence that the described procedure occurs in practice. Actually, it would reduce transparency of the project funding and may, therefore, gain little support in the process of project approval.

214 The projects with an estimated incremental domestic benefit that I presented above represent an exemption from this procedure. Moreover, it is not clear whether the derived figures describe the complete incremental domestic benefit.

Considering the sample of *197 projects* for which documents decribe the financing of the increment by financiers consistently, it turns out that in *64 projects* (32 percent), the GEF grant equals the size of the increment exactly, i.e., neither domestic actors nor international donors contribute to the financing of the incremental activities. For another nine projects, the GEF grant plus the official multilateral and unilateral payments cover the total increment. In the remaining *124 projects* (63 percent), financiers that I can clearly assign to the resource country participate in the financing of the increment.

Gross versus Net Incremental Cost: A Discussion

These observations relate to the theoretical discussion on the gross and net incremental cost in Section 4.3.1. The complete financing of the increment by the GEF implies that the pure form of the gross incremental cost principle is applied. However, based upon the information in the project documents, it cannot be concluded whether the absence of domestic financiers is attributed to the fact that an incremental domestic benefit is not taken into account in the incremental cost assessment or whether a positive benefit is simply not generated. In the latter case, the GEF funding is not related to the gross incremental cost as discussed before but refers to the straightforward case where the GEF funding covers incremental expenditures. Since, in both cases, the observable flow of GEF funding is the same, evidence of the existence and size of domestic benefits cannot be derived in this way.

In general, there is, on the one hand, variation in the extent to which the incremental domestic benefit is identifable or not and to which it is taken into account in the incremental cost assessment or not; on the other hand, the project documents show that there is also variation in the share of the increment that is financed by the GEF. Both characteristics can be combined in a different way. For each scenario described in Table 20, I make certain assumptions that I believe are not too unrealistic, in order to provide some general conclusions regarding cost-effectiveness and the distribution of the cooperation surplus.

In GEF (1996), it is implied that incremental activities may generate a domestic benefit that is unquantifiable or "considered as unimportant." In this vein, let us assume for simplicity that there is some positive contribution from these activities to the well-being of the resource country and in order to guarantee the incentive compatibility for its participation.[215] In other words, I assume that a

[215] Barrett (2002) argues that if incentive compatible transfers are provided to every resource country, the total provision of the global public good increases, with the consequence that each resource country benefits as member of the international community. As implied in GEF (1996), the domestic incremental benefit generated in this way is not deducted. More generally, following the noncooperative game theory (see

Table 20:
Summary of Allocative and Distributional Effects for Different Types of Incremental Cost Assessments

Incremental domestic benefit (DB) for the resource country (RC)			GEF grant	
Actual DB	Description of DB	DB considered in ICA	100% of increment	Less than 100% of increment
Zero (DB = 0)	No need to describe DB.	No need to consider DB.	Close to cost-effectiveness. No surplus for RC.	(Not defined.)[b]
Positive (DB > 0)	Identifiable. Described.	Fully considered.[a]	Not close to cost-effectiveness. Maximal surplus that is attainable for RC. (Gross incremental cost)	Close to cost-effectiveness.[b] No surplus for RC. (Net incremental cost)
	Not (fully) identifiable. Not (fully) described.	Partly considered.[a]	*Not close to cost-effectiveness. Maximal surplus that is attainable for RC. (Gross incremental cost)*	*Not close to cost-effectiveness. Positive surplus for RC.*
	Not (fully) identifiable. Not described.	Not considered.[a]	*Not close to cost-effectiveness. Maximal surplus that is attainable for RC. (Gross incremental cost)*	(Not defined.)[b]

[a]Only applies if GEF grant amounts to less than 100 percent of the increment. — [b]Other international grant funding is assumed to be zero.

project that yields no additional benefit to the resource country only represents a theoretical case and can, therefore, be excluded from further analysis, i.e., the cell with zero incremental domestic benefit in the column to the left is blocked out.

In the case of a positive incremental domestic benefit, it is shown that the 100 percent financing of the increment always corresponds to the gross incremental cost approach, which does not represent the cost-effective use of the GEF funding, and which is the most favorable policy for the resource country, since any domestic incremental benefit is not deducted from the external funding it receives.

Turning to the second column with the incomplete financing of the increment through the GEF, let us suppose for a moment that international grant funding other than GEF funding is zero. Consequently, the column illustrates cases where

Section 4.4.2), each project that is implemented in one country increases the incentive for other countries to accept an offer for the international financing of incremental activities (Barrett 2002).

funding for the increment is only mobilized by the GEF and domestic donors. This, in turn, implies that some extra domestic benefit is generated.

With respect to this incremental domestic benefit, two theoretical cases can be distinguished. Either the benefits are fully considered in the incremental cost assessment and, thus, fully deducted from the incremental expenditures or they are only partly considered. In the former case, the GEF funding is, in principle, used in a cost-effective way, since they only finance the global environmental spillovers. The resource country does not improve its well-being because the gross additional benefit it receives does not exceed the total costs of the increment net of the GEF grant. In the latter case, the parties divide the cooperation surplus among each other. Less emphasis is put on cost-effectiveness.

Given the often elusive nature of domestic benefits, for example, if local ecosystem services are involved that generate nonmarket values, the question is whether, in practice, all benefits can indeed be identified, described, and taken into account in the incremental cost assessment. Moreover, it is reasonable to assume that a positive incremental domestic benefit is only imperfectly described. Consequently, the second row in the table with a positive and fully considered benefit is also blocked out.

Accordingly, the remaining three cells in italics apparently describe the cases of empirical significance. Given the information provided in the official documents, it is not possible to allocate the 197 projects precisely across these cells. I can only use the identified frequency of projects with a 100 percent GEF share (32 percent) and those with a lower share and assign both figures as relative weights to the columns in the submatrix.

The fact that the cell in the right-hand column applies to two thirds of the cases confirms the theoretical finding by Cervigni (1998) that for the application of the incremental cost principle, the parties involved accept a *compromise* between the pure net and gross incremental cost approach. Otherwise, in one third of the cases, there is an agreement on the incremental cost assessment that is most favorable for the resource country.

I should mention two *caveats*. First, these findings rely upon figures for the average financing shares of the different donors. The variance in the size of the incremental cost among the projects is not considered. Based upon the results on the total project cost in Section 4.2.3.3, it can be expected that in the case of complete financing by the GEF, the increment will show a comparatively small size. This would imply that a gross incremental cost approach is actually applied in less than two thirds of the total activities funded by the GEF.

Second, the findings rely upon the assumption that there are no international grants, i.e., all observable international funding except from the GEF has the character of loan payments that have to be repaid by the resource country. If this assumption is relaxed, there is the basic problem of how to define the cost-effective use of GEF funding in the interplay with other international grants. When the definition of cost-effectiveness is narrowed to the question as to whether an incremental domestic benefit is considered in the incremental cost assessment or not, financial resources from the GEF are clearly not used effectively in the situations described by the two cells in the right-hand column: if the GEF grant and other international grants together finance 100 percent of the increment, the gross incremental cost approach illustrated in the cells to the left also applies in this case.

Bargaining on How to Account Incremental Domestic Benefits

The theoretical studies in Section 4.3.2 argue that taking incremental domestic benefits into account is the subject of a bargaining process between the resource country and the international donors. When applying this framework to the GEF projects, it is implied that the allocation of *bargaining power* influences the agreed upon level of conservation and the allocation of GEF funding, i.e., bargaining power matters for cost-effectiveness and the distribution of the cooperation surplus.

Regarding the bargaining among stakeholders in the process of designing GEF project proposals, information, such as discussions on alternative designs of project actions with different levels of conservation and the demand for finance, is not represented in the official project documents. The bargaining issue, therefore, does not seem to be accessible for an empirical analysis. However, given the average shares for the financing of the increment described in Figure 21, I ask whether different shares can be observed for *subsets of projects* that are *implemented in specific resource countries*.

Differences in the average share of the GEF or the sum of the shares of the GEF and international donors may imply that taking domestic benefits into account is systematically biased towards a specific group of resource countries. Any bias observed in this regard may be attributed, among other political factors, to the allocation of bargaining power.

As in Section 4.2.2, I consider projects in *LMMCs* and projects in *countries of different income classes*. I use the shares for each group of financiers described in Figure 21 as a benchmark. I indicate them in the second column of Table 21. The shares in the columns to the right refer to the different subgroups of resource countries. It turns out that there is only little variance in the shares for the GEF and the other international donors. Only projects in low income countries receive above-average support from both the GEF and the other official international donors. The average share of the domestic public sector in these countries is below average.

These findings illustrate that both the resource country's abundance of biodiversity and its income level do not have a constant influence on the agreed

Table 21:
Financing of the Increment in Various Groups of Resource Countries

Countries	All countries	Biodiversity	Income per capita		
		LMMCs	Low income countries	Lower middle income countries	Upper middle income countries
Number of projects	197	61	77	76	29
GEF	68	67	74	67	70
Other multilateral	4	1	6	2	0
Official unilateral	6	5	5	7	5
Domestic public sector	11	12	7	13	13
Local communities	2	3	2	2	1
Private sector	9	12	7	10	11

Note: Number of projects in absolute numbers. Other figures represent average financing in percent of the increment by financiers.

Source: Selected GEF project documents, own calculations.

upon finance of the incremental activities. Regarding the figures for low income countries, it needs to be studied further as to whether the strong commitment of the international donors is the result of the strong bargaining position of the low income countries threatening to excessively deplete their natural resource or is due to other political reasons.

Finally, the theoretical studies by Mohr (1990) and Sandler (1993) argue that the bargaining power of resource countries can find its expression in a current refusal to cooperate with international donors in order to make them offer larger payments for biodiversity conservation in the future. Such strategic behavior is only a credible option for a resource country if it can credibly threaten to deplete its own natural resources, i.e., it does not receive net domestic benefits from preserving the natural resource stock. My empirical analysis cannot contribute to this theoretical hypothesis, since it relies upon information concerning drafts of project contracts that have actually been concluded. Information regarding failed cooperation or canceled project proposals is not available.

The GEF's Role in Catalyzing Additional International Funding

In addition to the discussion on the incremental domestic benefit, the data provided in the incremental cost analyses can give some evidence of a complementary relationship between the GEF funding and funding from other donors. In this context, it is often assumed that the participation of the GEF in project financing leverages extra funding particularly from international sources. In other words, the emergence of the GEF as a funding institution is supposed to *catalyze additional funds* for investment in biodiversity conservation.

When assuming that the funding of the baseline is generally provided independently of the GEF support, the previous findings provide evidence of the size of the extra funding that is catalyzed.[216] More specifically, these are the findings on the aggregate cost of baseline and increment activities included in the study sample ($5.85 billion, $1.97 billion) and the shares in the baseline and the increment illustrated in Figure 21.

First of all, since baseline activities are approximately three times the size of the increment, more funding from unilateral and other multilateral donors in absolute terms is granted to baseline activities than to incremental activities. This implies that substantial international financing is provided independently of the GEF support. The actual proportion may even be higher, since funding that is not addressed in baseline activities is not taken into account in the aggregate figures.

According to its own indications for the period from 1991 to 2004, the GEF has mobilized $3.80 billon for project cofinancing for the entire biodiversity portfolio. Considering the 219 projects identifed, I estimate that my sample comprises $0.63 billion (=$1.97 billion × (1–0.68)) of this cofinancing.[217]

The shares in Figure 21 suggest that, on average, most of the extra funding is provided by the domestic public sector and the private sector. Comparing the financing of the baseline and the increment, the question arises as to how the shares differ among the donors (other than the GEF). This can provide some evidence of the importance of the GEF for the mobilization of funds by financier groups. For this purpose, I normalize the mean shares described in the increment, i.e., they are divided by 1–0.68. From a comparison of the normalized shares with the baseline shares, it turns out that aside from the private sector, local communities, and official unilateral donors show a relatively increased share, i.e., these donors seem to react more strongly to the GEF support.[218]

Finally, the question of international funding catalyzed by the GEF also relates to the question as to whether the financial resources provided are *new and additional* (CBD Art. 20(2)). Given the information in the official project documents, I cannot identify whether the funding for the GEF projects indeed re-

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216 To be precise, as noted in GEF (1996), the baseline may include activities that "have not yet attained a secured funding. " Thus, it cannot be ruled out that the GEF funding also catalyzes some funding for the baseline activities. Since, however, this does not seem to be the regular case, the values for the increment in Figure 21 understate the size of catalyzed resources only to a small extent.

217 Note that 0.68 is the average share of the GEF in the financing of the increment (Figure 21).

218 Again, it is difficult to verify the extent to which these resources actually represent grants or loan payments that finance incremental domestic benefits.

presents new financial resources or whether it is made available by reallocating existing flows of resources that address other aspects of official aid.

4.4 The Finance of the Global Environment Facility (GEF): Free Riding and Equitable Burden Sharing

While the previous section has analyzed how domestic biodiversity conservation, particularly protected areas, is financed by the international GEF as a representative of the rest of the world, I now study how the GEF as an international funding institution is endowed with financial resources by the donor countries. Generally, GEF funding granted to biodiversity conservation projects as well as to projects in the other GEF focal areas aims at safeguarding the environmental resources in the developing world that are of global importance. The spillovers that these resources generate are typically nonrival and nonexclusive in their use. The public good properties of *biodiversity* and ecosystem services are described in Section 4.1.1.

Furthermore, the abatement of anthropogenic greenhouse gas emissions, which contribute to *climate change*, and/or the abatement of *chlorofluorocarbon* and *persistent organic pollutants* is supposed to contain climatic and other atmospheric changes and thereby mitigate various adverse impacts upon humans and wildlife. No country can be excluded from a reduced risk of adverse impacts and the risk reduction for one country is unaffected by the number of the remaining countries. Consequently, nonrivalry and nonexclusiveness prevail for the benefits of abatement (Murdoch and Sandler 1997).

Finally, environmental protection in developing countries helps to preserve their economic capacities from irreversible degradation as is expressed, for example, in *desertification*, soil erosion, or eutrophication. Although this kind of protection contributes to local public goods, it can—similar to foreign aid—serve to mitigate several negative spillovers or global public bads, such as infectious diseases, war, crime, or illegal migration (Jayaraman and Kanbur 1999).

While the GEF funding contributes to the provision of environmental protection, the GEF as an international regime can itself be considered a public good: as an *intermediate global public good*, the GEF assists the provision of the *final global public goods* described (Kaul et al. 1999). The actual level of global environmental protection is, inter alia, explained by the size of the multilateral GEF trust fund and the additional unilateral transfers.

Regarding the challenges in global environmental policy, the literature often notes that the GEF fund has previously been only *modest in size* (Connolly 1996; Fairman 1996). This appraisal supports the (commonly agreed upon) conclusion that the total of current international transfers falls short of the amount needed to safeguard biodiversity and other environmental resources to a globally efficient extent.

Several factors, such as the lack of institutions that enable coordination between donors and the developing country, or distortions in domestic policy that create perverse incentives for nonsustainable resource use, impede both multilateral and unilateral efforts in the financing of environmental protection including biodiversity. These factors diminish the effectiveness of the international assistance and may reduce the willingness of international donors to provide funding. In addition, given the governance structure of the GEF as a multilateral institution, donors may be unwilling to provide funds specifically to the GEF because they have less control over the allocation and use of their resources as compared to unilateral assistance (Nunnenkamp 1992; see also Section 4.1.1.3).

In spite of its modest financial size, the GEF serves as the financial mechanism for major international environmental agreements (IEAs) and, thus, plays a prominent role in international policy. Therefore, the financing of the GEF requires closer consideration. In the political debate, there have been repeated calls for the *financial strengthening* of the GEF (Wolfensohn 2004; Perrings and Gadgil 2003: 547). These calls for increasing contributions to the GEF trust fund implicitly assume that the current endowment of the GEF is suboptimally low. Suboptimality can stem from factors that are not due to the political and institutional environment in the resource countries and that have not been mentioned before. These are:

- The problem of *incomplete information* within the donor community. Donor countries are unaware of, or underestimate the importance of, global biodiversity and global environmental protection for the well-being and inappropriately prioritize other domestic or international public goods.
- The problem of *free riding*. The services generated from activities that receive GEF funding provide benefits to a large group of countries, whereas the benefits are typically nonrival and nonexcludable. Not all of these countries contribute to the finance of the GEF and it is generally unclear as to whether a country's contribution corresponds to the size of the benefit it receives in each individual case.

The following analysis concentrates upon the issue of free riding. Free riding in the provision of environmental protection as a global public good suggests that donors devote less than the optimal amount of financial resources to this purpose. The major questions in this context are

- whether empirical evidence can support the hypothesis of free riding behavior among (potential) GEF donors, and

- whether the institutional environment of the financing of the GEF can be improved in order to increase the future endowments of the GEF fund.

In Section 4.4.1, I describe the arrangement between GEF donors and provide figures on the size of the GEF trust fund. Based upon this, I introduce selected models of collective action theory and noncooperative game theory whose findings help to explain the observable interactions between donor countries in the finance of the GEF (Section 4.4.2). Finally, I use the methods of collective action theory to conduct an empirical analysis of burden sharing in the GEF finance (Section 4.4.3). I investigate to what extent equity concepts are satisfied and what the empirical results on equity imply with regard to efficiency.

4.4.1 Generating Funds: An Empirical Description of the GEF Replenishment Process

The GEF mechanism relies upon a multilateral trust fund located at the World Bank (see Section 4.2.1). Since its disbursements are made upon a grant basis, the fund needs to be replenished from time to time. In its pilot phase, the fund was endowed from various sources with approximately *$1 billion* (Pearce 1995: 139ff.). Since 1994, there have been regular replenishments of the fund over a four-year period. Donors to the GEF are expected to have agreed on a fourth replenishment while this study is in press. Yet the financing of the GEF-4 is outside the scope of my analysis.

The GEF replenishment process generally follows several steps. First, in order to determine the total size of replenishment, the donor involved and the resource countries estimate the financial requirements that are programmed for in the different focal areas of the GEF. Based upon this, they negotiate the *target size* of the replenishment (GEF 2002a). In this respect, the parties intend to ensure adequate funding for the defined objectives of environmental protection. At the same time, they aspire towards a fair burden-sharing arrangement that is guided by the principles of "transparency, equity and ability to pay" (GEF 2001a).

Certain (minimum) requirements have an influence upon the size of the individual contributions to the GEF, although each donor country can, in principle, control the size of its individual commitment. Donors are distinguished as *nonrecipient* countries, i.e., (developed) countries, which do not receive project grants from the GEF, and *recipient* countries, which participate in the replenishment but are still eligible to receive GEF grants to finance projects within their territory. The countries of the OECD Development Assistance Committee (DAC) represent the group of nonrecipient countries.

When making a commitment to the replenishment process, donor countries of both groups must satisfy the *minimum contribution requirement*. This requirement is set at the amount of 4 million in SDR (special drawing rights) (about $5 million).

In order to reach the target size of the replenishment, the individual contributions of the nonrecipient countries as the major donors are determined according to a burden-sharing framework that defines individual *basic shares* for each donor (GEF 2001a). For the first replenishment process, these shares were borrowed from the financing scheme of the International Development Association (IDA) (GEF 2004b). These shares also served as points of reference in subsequent replenishment processes. To derive the *basic contribution* for each of the nonrecipient donors in absolute terms, the individual shares are related to the target size.

Aside from the basic contributions, some nonrecipient donors make *supplementary contributions*. This holds for donors with a small basic share in order to satisfy the contribution requirement or for some major donor countries that adjust the total contributions towards the full funding of the target size. However, in addition to this, both nonrecipient and recipient counties are allowed to make *additional contributions*.

In the previous replenishment processes, a different number of countries have joined in as contributing participants. According to the GEF (2002c, 2003, 2004b), all of the 22 OECD DAC countries plus the Republic of Korea have taken part in all of the previous processes as nonrecipient countries.[219] Among the recipient countries, seven of them have made commitments to all three replenishments (China, Cote d'Ivoire, Czech Republic, India, Mexico, Pakistan, and Turkey). Bangladesh and the Slovak Republic participated only in the first replenishment, and the Russian Federation only in the second one. Argentina, Brazil, and Egypt withdrew after the second replenishment. Nigeria and Slovenia have joined the donor community since.[220]

For the GEF-1 replenishment, only 18 out of 35 donor countries contributed an amount in excess of the minimum contribution requirement. In the second replenishment, 19 out of 36 donors made contributions in excess of this requirement. In the third replenishment, it was 24 out of 32 countries.[221]

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[219] This group of countries is identical to the *high income OECD members* (excluding Iceland) in the World Bank country classification.

[220] I primarily use data from GEF (2004b). I consider certain additions that are not indicated in this source but that are presented in GEF (2002c, 2003).

[221] Some countries, like Bangladesh and Slovenia, have made contributions of less than the minimum requirement. The reasons for this have not been explained in the official documents.

Considering the supplementary contributions, 10 to 12 donors representing both nonrecipient and recipient countries made use of this option in each of the three replenishments. In total, these contributions add between 1.7 and 2.7 percent to the total of the (adjusted) basic contributions. For the major donor countries, these contributions have usually amounted to a size that is rather small compared to that of the individual basic contribution.[222]

Donors in a negotiation process seem to take into account several objectives simultaneously. In particular, there is a likely interplay between the determination of the target size of new pledges and the maintenance of the previously agreed upon burden-sharing regime. For example, Sjöberg (1999) reports that, during the second replenishment process, the United States had difficulties in mobilizing financial resources. Meanwhile, other donors, such as France, insisted on making contributions contingent upon the US share.

In the past, the sum of all contributions committed ex ante did not always correspond exactly to the total of the financial resources actually provided. For special reasons, such as budgetary constraints, a donor country is allowed to deviate temporarily from its commitments, i.e., it may provide less than the committed amount within the considered GEF period. However, nonaccomplished commitments are considered outstanding arrears, meaning that a donor is requested to make up for the arrears in the following replenishments (GEF 2002c, 2002d).[223] Furthermore, donors possess some flexibility with respect to the time schedule and the currency in which they provide their contribution. Since inflation in local currencies and exchange rate fluctuations can lead to a reduction in effective contributions, donors are requested to preserve the present values of their commitment when making their payments to the GEF trust fund (GEF 2001e, 2001f).

According to the GEF (2005d), donors provided the amount of *$2.0 billion* for GEF-1. In 1998, new pledges of $2.0 billion were committed. Together with unallocated means from GEF-1, the replenishment for GEF-2 amounted to *$2.8 billion*. In 2002, the donors agreed to provide resources for a third replenishment of *$3.0 billion* (GEF 2005d; UNEP 2002). These figures imply that, in current dollars, the total size of the GEF fund has increased steadily across the various replenishment processes.

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222 The United Kingdom committed supplementary contributions to the GEF-3 replenishment at the size of about 15 percent of its basic contribution. I derive similar figures for Denmark and Sweden. All other nonrecipient countries either made supplementary contributions of a smaller relative size or no contributions at all.

223 The United States only paid 60 percent of their commitment in the GEF-2 replenishment but increased their commitment in the GEF-3 replenishment by more than 16 percent. In another case, the arrears of Argentina and Egypt in the GEF-1 replenishment were cleared after negotiations with the GEF trustee (GEF 2003).

Table 22:
Contributions of Nonrecipient Countries to GEF Replenishments, in Current Dollars

Contributing participant	GEF-1	GEF-2	GEF-3
Australia	29.20	32.21	35.00
Austria	20.01	20.17	22.44
Belgium	22.86	34.20	41.80
Canada	86.55	88.24	102.58
Denmark	35.14	28.68	35.44
Finland	21.65	22.05	26.55
France	143.27	144.83	163.35
Germany	239.99	220.00	263.67
Greece	5.00	5.49	5.71
Ireland	2.40	5.49	5.71
Italy	114.69	90.53	105.22
Japan	414.63	412.60	422.72
Rep. of Korea	5.60	5.49	5.52
Luxembourg	5.60	5.49	5.07
Netherlands	71.41	72.80	79.10
New Zealand	5.60	5.49	5.07
Norway	30.72	31.33	25.31
Portugal	5.60	5.49	5.07
Spain	17.32	16.51	19.17
Sweden	58.28	57.80	72.24
Switzerland	44.79	43.87	58.25
United Kingdom	134.55	138.91	190.07
United States	430.00	430.00	500.00

Source: GEF (2004), GEF (2002c, 2003) for Belgium, Greece, Rep. of Korea, and Luxembourg with respect to GEF-1.

Table 22 presents the contributions of the *nonrecipient countries* in current dollars.[224] Comparing the individual contributions across the various replenishments, there are only slight variations in the contributions and, therefore the shares of the major donor countries remain rather constant, which is apparently a result of the applied basic share framework. Additional contributions by certain donor countries do not lead to significant differences over time.

Furthermore, it turns out that contributions by the G-7 countries, i.e., the seven major industrialized countries, together comprise approximately 75 percent of total funds provided to the GEF in each replenishment process. For reasons of

[224] I converted the figures given in GEF (2002c, 2003, 2004) from local currencies and SDR into current dollars by using the dollar-SDR exchange rates indicated in these documents.

space, the contributions by the recipient countries are omitted here (GEF 2002c, 2003, 2004).

Given the empirical facts on the finance of the GEF, I continue with an analytical analysis of the interactions between the donor countries.

4.4.2 Interactions of Donor Countries from a Theoretical Perspective

Since the donor countries have private information concerning the benefits of global environmental protection each of them actually receives and since there are difficulties in detecting free riding, a hypothesis of suboptimality due to free riding rests on a suggestive conclusion. However, this hypothesis has not been tested empirically. Similarly, the actual impact of free riding is also unclear, i.e., it is unknown how many financial resources would be provided if donors contributed according to their true valuation of the benefits received.

In the literature, free riding in the context of the private provision of a (global) public good other than environmental protection is empirically analyzed using econometric methods (Khanna 1993; Sandler and Murdoch 1990). However, the application of these methods to the financing of the GEF is limited by the small data set for the GEF replenishments.[225] For this reason, I limit my further analysis to qualitative statements that are derived from a comparison of empirical information on the GEF replenishment process (Section 4.4.1) and theoretical findings from both collective action theory (Section 4.4.2.1) and noncooperative game theory (Section 4.4.2.2).[226] Given the nonexclusive spillovers from environmental protection, the key issue is whether the community of countries manages to implement effective negotiation rules and design intelligent agreements that deter free-riding behavior and, thus, guarantee sufficient financial resources for environmental protection in the developing countries.

4.4.2.1 Collective Action and the Finance of Global Environmental Protection

I initially use the tools of collective action theory to study the interactions between countries that contribute to the GEF trust fund. Collective action theory

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[225] Especially the time series of the relevant variables are too short. This data set, which consists of about 35 donors participating in the various replenishments, is not sufficient to perform two-stage least square estimations, as do the other studies on private public good provision (Khanna 1993; Sandler and Murdoch 1990).

[226] For reasons of space, I cannot deal with the entire literature concerning this issue. I place the focus upon selected models in collective action theory and noncooperative game theory. However, I recognize that other papers that I do not discuss also give further valuable insights.

represents an adaptation of the public good theory to organizational cooperation between economic actors that are sovereign in their decision making. In the literature, this theory is applied to domestic collective goods and interactions between private actors, as well as to international collective goods and interactions between sovereign countries (Sandler and Hartley 2001). These interactions were first conceptualized theoretically in the standard model of the private or voluntary provision of a single pure public good (Olson 1965).[227]

The Classic Model of the Private Provision of a Pure Public Good

There are n countries; each of them allocates its national income, Y_i, $i = 1..N$, to purchases of a private consumption good, c_i, and the supply of a pure public good, g_i. Individual supplies of the countries add up to form the total supply of the public good, G, i.e., $G = \sum_i g_i$. In other words, a summation technology with respect to G is assumed. Due to the additivity, the individual contributions are perfectly substitutable.

The utility of the individual country is described by a continuous, strictly quasi-concave utility function, u_i. Both the private and public goods are normal with positive income elasticity. Each country then maximizes its utility by allocating its income between c_i and g_i. In the standard model, each country believes that the contributions by the others are independent of its own decision, i.e., Nash–Cournot behavior is assumed (Bergstrom et al. 1986; Cornes et al. 1999).

This model framework is appropriate to explain the interactions between donor countries in the GEF replenishment process whenever the model's assumptions represent the actual process in a suitable way. This is, first, that countries are represented by governments that decide on the size of the contributions as unitary actors. Donor countries share a single global public good that I assume is represented by the range of the external benefits generated from environmental protection in developing countries. The supply of this final public good is approximated by the total size of the GEF funding granted to these countries.[228]

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[227] Formal representations of this model are, inter alia, given in Andreoni (1988), Bernheim (1986), Bergstrom et al. (1986), and Cornes and Sandler (1984). While the model in its original form considers interactions between consumers, it is also applied to the international level to describe interactions between sovereign countries (Inori 1996; Jack 1991; Kemp 1984).

[228] A more precise way of modeling is to define the public good as B, and conservation costs as $C(B)$ with $dC/dB > 0, d^2C/dB^2 > 0$. When assuming that the GEF funds are used in a cost-effective way, and developing countries are only compensated for

In this respect, individual contributions are also additive and perfectly substitutable. Finally, whether the assumption that each donor country believes the contributions by the others are independent of its own actions (Nash–Cournot behavior) is justified is discussed below. For the moment, I regard Nash–Cournot behavior as a reasonable assumption.

In order to derive the Nash equilibrium for some specific functional forms, I adapt the simple *two-country framework* by Itaya and Yamada (2004). The countries display identical log-linear preferences but different levels of national income. The two countries are indexed with i and j, where $Y_i > Y_j$. The utility functions of the two countries are given by

$$(4.18)\quad u_i(c_i, G) = \alpha \log c_i + (1-\alpha)\log G\,, \quad u_j(c_j, G) = \alpha \log c_j + (1-\alpha)\log G\,.$$

The budget constraint for each country is

$$(4.19)\quad Y_i = c_i + g_i\,, \quad Y_j = c_j + g_j\,.$$

The quantity of the public good, i.e., the size of the GEF trust fund, is defined as

$$(4.20)\quad G = g_i + g_j\,.$$

Contributions are defined in monetary units. In the literature, they are sometimes defined alternatively in physical units that each country purchases at a given price per unit.

To evaluate noncooperative collective actions, first the *Pareto-optimal* allocation is described. The conditions for Pareto efficiency are essentially described by the *Bowen–Lindahl–Samuelson condition*, saying that in the optimum the sum of the marginal rates of substitution between the collective/public good and the private good is equal to the marginal rate of transformation in production (Laffont 1988: 37). In case of a summation technology in the public good provision, as I assume, the marginal rate of transformation is equal to one (Varian 1992: 419):

$$(4.21)\quad \frac{du_i}{dG}\bigg/\frac{du_i}{dc_i} + \frac{du_j}{dG}\bigg/\frac{du_j}{dc_j} = 1\,.$$

their net incremental conservations costs, it holds $\sum_i g_i = G = C(B)$ (Barrett 1994a). Solving for B, a utility function, $\tilde{u}_i(c_i, B(G))$, can be defined that also satisfies strict quasi-concavity, i.e., $\tilde{u}_i(c_i, B(G)) \equiv u_i(c_i, G)$.

When substituting G using equation (4.20) and c_i and c_j using the budget constraints, the optimality condition transforms to

$$(4.22)\quad g_i^* + g_j^* = (1-\alpha)(Y_i + Y_j).$$

This equation implies that there is no unique Pareto optimum, but that a range of allocations exists that satisfies efficiency (Varian 1992: 419).

To select a social optimum from these Pareto-efficient allocations, I assume a simple welfare function that is linear in the utilities of the two countries. The weight for country i in the welfare function is denoted by λ, the weight for country j is $1-\lambda$ accordingly. The welfare function is, then, maximized subject to the specified Bowen–Lindahl–Samuelson condition, i.e.,

$$(4.23)\quad \max_{g_i \cdot g_j \geq 0} \lambda u_i(g_i, g_j, Y_i) + (1-\lambda) u_j(g_i, g_j, Y_j)$$

$$s.t.\ g_i + g_j = (1-\alpha)(Y_i + Y_j).$$

It can be shown that the welfare function is increasing and quasi-concave in individual contributions. Since the efficiency constraint is linear in the space of individual contributions, a unique social optimum exists.

Based on equation (4.23), I represent the optimization problem by a Lagrangian function $L(g_i, g_j, \eta)$, where μ denotes the Lagrangian multiplier. The partial derivative of the Lagrangian with respect to g_i and g_j is

$$(4.24)\quad \frac{dL}{dg_i} = -\frac{\lambda\alpha}{Y_i - g_i} + \frac{\lambda(1-\alpha)}{g_i + g_j} + \frac{(1-\lambda)(1-\alpha)}{g_i + g_j} + \mu = 0,$$

$$\frac{dL}{dg_j} = -\frac{(1-\lambda)\alpha}{Y_j - g_j} + \frac{\lambda(1-\alpha)}{g_i + g_j} + \frac{(1-\lambda)(1-\alpha)}{g_i + g_j} + \mu = 0.$$

Given these results and the condition of Pareto efficiency, the social optimal contributions are

$$(4.25)\quad g_i^* = Y_i - \lambda\alpha(Y_i + Y_j), \quad g_j^* = Y_j - (1-\lambda)\alpha(Y_i + Y_j).$$

When returning to *private allocation*, the maximization problem for country i is described as follows (the maximization for country j can be stated accordingly):

$$(4.26)\quad \max_{g_i \geq 0} u_i(g_i, g_j) = \alpha \log(Y_i - g_i) + (1-\alpha)\log(g_i + g_j).$$

Each of the two countries chooses a nonnegative contribution given the choice of its opponent. The first-order conditions for country *i* serve to determine the optimal reaction path for country *i*:

$$(4.27)\quad \frac{du_i}{dg_i}=0 \Leftrightarrow g_i^*(g_j)=(1-\alpha)Y_i-\alpha g_j .$$

Due to the assumption of identical preferences, the reaction path for country *j* is symmetric, i.e., only the indexes *i* and *j* in equation (4.27) are exchanged. I use both reaction paths to solve the Nash equilibrium:

$$(4.28)\quad g_i^*=\frac{Y_i-\alpha Y_j}{(1+\alpha)},\ g_j^*=\frac{Y_j-\alpha Y_i}{(1+\alpha)} .$$

For an interior solution, the sum of the individual contributions describes the equilibrium level of the public good:

$$(4.29)\quad G^*=g_i^*+g_j^*=\frac{(1-\alpha)}{(1+\alpha)}(Y_i+Y_j).$$ [229]

By comparing equation (4.29) with equation (4.22), it can be seen that the public good is undersupplied in the Nash equilibrium. At least one country relies upon the contribution of its opponent and free rides.

For the *n-country* model with identical preferences in its generic form, Bergstrom et al. (1986) show that, in equilibrium, the countries are divided into two subgroups: countries with an income below a certain critical value, Y', make no contribution, while each country with an income exceeding Y' spends Y_i-Y' on the supply of the public good. Thus, the total supply is the sum of individual incomes in excess of Y'. Noncontributing countries spend their entire income on the private good, whereas each contributor spends just the amount of Y'. I can demonstrate that this finding of identical expenditures for consumption also applies to the two-country case with the functional forms introduced above.

Finally, I calculate the share, μ, of each country in the total finance:

$$(4.30)\quad \mu_i=\frac{g_i^*}{G^*}=\frac{Y_i-\alpha Y_j}{(1-\alpha)(Y_i+Y_j)},\ \mu_j=\frac{g_j^*}{G^*}=\frac{Y_j-\alpha Y_i}{(1-\alpha)(Y_i+Y_j)} .$$

Nash–Cournot behavior in the GEF context would imply that there is indeed free riding among GEF donors. In order to increase the size of the GEF fund,

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229 For $Y_i>Y_j>0$, it turns out that a unique interior equilibrium exists whenever $\alpha Y_i<Y_j$.

policy measures that arrange institutional change and, thereby, limit the extent of free riding have to be considered.[230]

Drawing upon the empirical description in the previous section, there is no direct support for a Nash–Cournot behavior in negotiations over a GEF replenishment. Moreover, the applied scheme of the basic shares and, particularly, the anecdote about the pegging of the French contribution to the size of the US contribution seems to indicate that (major) donor countries do realize that the contributions of others are not independent of their own decisions (see Sjöberg 1999). In contrast, it cannot be observed that the committed supplementary contributions are based upon coordinated arrangements or that they follow any pre-agreed-upon institutional rule. Thus, Nash–Cournot behavior may explain the allocation within this subset of contributions.

Given the standard model of collective action outlined above, several studies demonstrate how changes in the model setting lead to various changes in the outcome, i.e., the total supply of the public good, as well as the composition of the group of donors and the size of their contributions. On the one hand, these changes relate to changes in parameter values. This includes (1) the number of countries involved (Andreoni 1988), (2) the income of some countries relative to that of others, including income redistribution (Bernheim 1986; Warr 1983), and (3) changes in the price of the public good in some countries relative to that in others (Bruce 1990; Ihori 1996).[231] For reasons of space, I omit the description of the results from these studies.

On the other hand, several studies analyze the impact of changes in the model structure. This refers to heterogeneous preferences as opposed to homogenous preferences for the countries (Andreoni 1986; Bergstrom et al. 1986; Warr 1983) but also to changes in behavioral assumptions and the additional introduction of a joint private good or multiple pure public goods. Considering the *alternative behavioral assumptions*, two major types are studied in the literature. These are the Stackelberg leader-follower relationship (Bruce 1990; Sandler 1992: 57f.) and interactions in a Lindahl process (Laffont 1988; Sandler and Murdoch 1990).

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230 Olsen and Zeckhauser (1966) discuss whether moral suasion is an effective instrument in this regard.

231 With respect to the provision prices, studies analyze whether there are possibilities to "trade" public goods (Jack 1991; Boyer 1988; Kemp 1984; Kiesling 1974): when the supply costs of an international public good differ between countries, they may "trade" in that they become specialized according to their comparative cost advantage. Analogously to trade with private goods, a country that receives spill-ins may provide transfer payments to increase the production of the public good in other countries instead of producing the public good on its own. Depending upon the technology for the public good provision, trade can alternatively refer to countries that purchase inputs abroad for the domestic production of the global public good.

By continuing with the functional forms introduced, I describe how a first-mover advantage of the *Stackelberg leader* influences the model outcome in comparison to the Nash–Cournot case. Subsequently, I discuss the Lindahl behavior. Finally, I conduct a qualitative discussion of joint products of public good provision and of the case of multiple public goods.

First-Mover Advantage

Suppose the high income country i is the leader, i.e., it can make a credible first move in contributions to the public good. The follower (the low income country), j, optimizes its choice given the decision by the leader. Technically, I substitute g_j in equation (4.23), the reaction path of the leader, i, with the reaction path of follower country, j, given in equation (4.25). Country i now chooses g_i in order to maximize its utility:

$$(4.31)\quad \max_{g_i \geq 0} u_i(g_i) = \alpha \log(Y_i - g_i) + (1-\alpha)\log\left(g_i + g_j^*(g_i)\right).$$

Since the second-order derivative of u_i with respect to g_i is negative for $g_i > 0$, the first-order condition yields the utility-maximizing contribution

$$(4.32)\quad \frac{du_i}{dg_i} = 0 \Leftrightarrow g_i^* = Y_i - \alpha\left(Y_i + Y_j\right).$$

Given g_i^* and the reaction path for country j, I can derive g_j^*, as well as the level of total contributions in an interior equilibrium, $(g_i^*, g_j^*) > 0$, i.e.:

$$(4.33)\quad g_j^* = Y_j - \alpha(1-\alpha)\left(Y_i + Y_j\right) \text{ and}$$

$$(4.34)\quad G^* = g_i^* + g_j^* = (1-\alpha)^2\left(Y_i + Y_j\right).^{232}$$

The comparison of G^* in equation (4.34) with the Pareto optimum described in equation (4.22) again reveals an undersupply of the public good.

The resulting shares, γ, of country i and j in the finance of the public good are

$$(4.35)\quad \gamma_i = \frac{g_i^*}{G^*} = (\alpha-1)^{-2}\frac{Y_i}{\left(Y_i + Y_j\right)} - \frac{\alpha}{(\alpha-1)^2} \text{ and}$$

232 It turns out that an interior Stackelberg solution requires that $(1-\alpha)Y_i > \alpha Y_j$ for $g_i^* > 0$ and that $(\alpha(1-\alpha))Y_i < (1-\alpha(1-\alpha))Y_j$ for $g_j^* > 0$.

$$(4.36)\quad \gamma_j = \frac{g_j^*}{G^*} = (\alpha-1)^{-2}\frac{Y_j}{(Y_i+Y_j)} - \frac{\alpha(1-\alpha)}{(\alpha-1)^2}.$$

I compare the Stackelberg outcome with the Nash–Cournot outcome. When all other things are held equal, the total supply of the public good is relatively smaller than the Nash–Cournot outcome. Furthermore, the high income country, *i*, makes a comparatively smaller contribution to the public good provision, while the contribution of the low income country, *j*, is generally comparatively larger (see Bruce 1990; Sandler 1992: 57f. for the generic case). When comparing the absolute shares in the total finance, it turns out that the leader country displays a larger share, i.e., $\gamma_i > \gamma_j$, whenever $Y_i > ((1+\alpha^2)/(1-\alpha^2))Y_j$.[233]

Based upon equations (4.32), (4.33), and (4.25), I determine the gap between individual contributions in the Stackelberg outcome and the social optimum. For the leader, the gap is $(1-\lambda)\alpha(Y_i+Y_j)$. For the follower, it is $(\lambda-\alpha)\alpha(Y_i+Y_j)$. These terms imply that the gap for leader is necessarily positive; it is larger the larger $1-\lambda$ is, which is the share of the follower in the welfare aggregation. A gap for the follower country does not necessarily exist: for $\lambda < \alpha$, the follower contributes more than required in the social optimum. More generally, the gap for leader is larger than the gap for the follower whenever $(1+\alpha)/2 > \lambda$. Consequently, for reasonable assumptions concerning the preferences and welfare aggregation, it turns out that the gap is larger in absolute terms for the high income country, *i*.

Considering the empirical description of the GEF replenishment process, the question is whether contributions by the nonrecipient countries are predetermined by the scheme of basic shares in a way that would support the leader-follower hypothesis. When the group of nonrecipient countries indeed behaves as a Stackelberg leader, i.e., these countries can make a credible first move in contributions, the participating recipient countries can only act as followers. When using the theoretical result on a shortage in contributions in the Stackelberg outcome, I cannot conclude which country should increase its contributions in order to approach the optimum, since there is range of alternative Pareto optima (see equation (4.22)). Only if I apply a welfare function that is linear in utilities do the results tend to imply that nonrecipient countries should increase their contribution to the GEF. An assessment of the contributions of the participating recipient countries depends upon the preference for protection and the moral assumption in the welfare aggregation.

This conclusion, however, rests upon the emphasis of the basic shares scheme. In contrast, this scheme does not determine the contributions of the nonrecipient

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233 Note that, for $\alpha = 0.6$, the scaling factor in the inequality is greater than 2; for $\alpha = 0.9$, it is about 10.

countries entirely; the target size of the replenishment also has an influence upon the absolute size. Since this target size results from negotiations in which recipient countries also participate, a potential first-mover advantage for the non-recipient countries is likely to be weakened in this respect.

To resume, additional information on negotiations in the replenishment process is required to support the leader-follower hypothesis. Furthermore, it has to be clarified as to the extent to which a donor that simultaneously represents a recipient country benefits from an increase in the individual contributions itself and, thus, to what extent this connection motivates its participation in a GEF replenishment process. There may be some sort of issue linkage between a recipient country's payments to the GEF and the environmental and development aid it receives from other international donors.[234]

Regarding the contribution for recipient countries, the empirical analysis in the previous section suggests that the size of the contribution is often dominated by the provisions in the minimum contribution requirement, although an increasing number of recipient countries make supplementary contributions in excess of this requirement.

If the aim is to increase the total contributions to the GEF and the leader-follower hypothesis can be maintained in spite of the caveats mentioned, the theoretical findings imply that the position of the recipient countries should be enhanced, and more generally, that the cooperative elements in the replenishment process should be strengthened.

A Lindahl Process

In contrast to noncooperative strategic interactions, let us assume that countries cooperate, i.e., they meet and exchange information on the total supply of the public good. To cover the supply costs, individual cost shares, θ_i with $\sum_i \theta_i = 1$, are determined, whereas the share of an individual country reflects its valuation of the public good relative to the valuation by the other countries. A Lindahl equilibrium is then characterized by a vector of shares, θ_i^*, $i = 1..n$, such that the utility of each country is maximized and each country consumes an identical amount of the public good (Laffont 1988; Sandler and Murdoch 1990).

Returning to the two-country model, I introduce endogenous Lindahl shares, θ_i with $\theta_i + \theta_j = 1$, for the financing of the public good. Consequently, the budget constraint for country i is defined as

234 For a discussion on the question of issue linkage, see Barrett (2001) and Carraro and Siniscalco (1993).

$$(4.37)\quad Y_i = c_i + g_i = c_i + \theta_i G_i .$$

Accordingly, country *i* maximizes its utility by choosing the desired bundle of private consumption, c_i, and quantity of the public good, G_i:

$$(4.38)\quad \max_{c_i \geq 0, G_i \geq 0} u_i(c_i, G_i) \quad s.t.\ c_i + \theta_i G_i = Y_i .$$

Solving the constrained maximization problems yields the share

$$(4.39)\quad \theta_i = (1-\alpha)\frac{Y_i}{G_i} .$$

Due to the symmetry assumption, the corresponding term for country *j* can be determined. By applying the conditions for a Lindahl equilibrium, i.e., $\theta_i + \theta_j = 1$ and $G_i = G_j = G^*$, the equilibrium level of the public good is derived as

$$(4.40)\quad G^* = (1-\alpha)(Y_i + Y_j) .$$

By substituting G^* for G_i in equation (4.39), the resulting equilibrium shares in the financing correspond to the individual income shares:

$$(4.41)\quad \theta_i{}^* = \frac{Y_i}{(Y_i + Y_j)},\quad \theta_j{}^* = \frac{Y_j}{(Y_i + Y_j)} .$$

Using these equilibrium terms, the individual contributions are described by

$$(4.42)\quad g_i{}^* = (1-\alpha)Y_i,\quad g_j{}^* = (1-\alpha)Y_j .$$

It can be seen that the Lindahl outcome in equation (4.42) satisfies the efficiency conditions described in equation (4.22). The impact of free riding is eliminated. This also implies that in the Lindahl equilibrium, the provided quantity of the public good is greater than in the noncooperative Nash–Cournot and Stackelberg outcomes. A comparison of equations (4.42) and (4.25) shows that the Lindahl equilibrium coincides with the social optimum introduced above whenever the derived Lindahl shares in equation (4.41) represent the utility weights in the welfare function, i.e., $\theta_i = \lambda$ and $\theta_j = 1 - \lambda$.

Considering the GEF, the question is whether negotiations in the replenishment process correspond to a theoretical model of a Lindahl process. When focusing upon the group of the nonrecipient countries only, the basic shares may be considered as analogues to Lindahl shares. Together with the fact that the definition of the target size of the replenishment is the first step in the negotiations, it can be hypothesized that interactions between those donor countries for which basic shares are defined represent a Lindahl process and that the sum of contributions by these countries in each of the previous replenishments represents a Lindahl equilibrium.

If a Lindahl equilibrium is attained, each donor country provides a Pareto-efficient contribution. Consequently, it would not be rational for any of them to increase its contribution. In this sense, the participating donors' willingness to pay is completely exhausted in the current replenishment arrangement. Other things held equal, it is not possible to reach an agreement on any further increase of the total contributions.

Nevertheless, the Lindahl equilibrium imposes strong assumptions with respect to the enforcement of the allocation that hardly hold in practice: when a Lindahl equilibrium is established, each country is aware of the external benefits its contribution has for the other countries and that there are no obstacles in revealing and observing the true benefit of each country. In other words, none of the countries possesses private information on its own valuation of the global public good—or, at least, none of them makes use of it.[235]

Another caveat is that the concept of the Lindahl equilibrium when applied to the finance of the GEF cannot explain supplementary contributions aside from the contributions that are derived from the basic shares. If the latter indeed corresponded to Lindahl shares, no country would have an incentive to increase its contribution further.

Joint Private Good

In addition to alternative behavioral characteristics of the participating donor countries, the literature discusses the case where additional private or public goods are connected with the global public good considered and analyzes the impact upon the quantities provided.

Let us consider the case where contributions to a (global) public good simultaneously generate a joint product, z_i, that is a private good, i.e., yields exclusive benefits for a specific donor. The studies in the literature typically

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[235] To put it differently, if a Lindahl equilibrium is established as a self-enforcing outcome, truthful revelation of the valuation is the dominant strategy for each of the participating countries. However, it is supposed that in many cases of (international) collective actions that are comparable to the finance of the GEF, revelation of the true benefits is indeed a problem. The theoretical research in this respect focuses upon the design of the mechanism for financing public goods under which the truthful revelation of valuations (benefits) is an "equilibrium" outcome. A mechanism that satisfies this requirement is the Clarke–Groves mechanism (Mas-Colell 1995: 876ff.; Siebert 2005: 89ff.).

describe the provision of the public good and the joint private good in fixed proportions and assume Nash–Cournot behavior for the countries. It turns out that the gap between Pareto-efficient contributions and contributions in the noncooperative outcome, other things held equal, decreases with an increasing proportion of the private good (Sandler and Hartley 2001).[236]

In other words, while free riding among contributing countries (donors) may not be avoided entirely, the incentive to free ride can be mitigated if, from the point of view of a donor, the financing of the pure public good yields a joint product that generates exclusive benefits. Considering the GEF, the supported projects explicitly address the protection of globally important environmental resources. Accordingly, it is hardly discernable as to how far the GEF-supported projects yield benefits that occur exclusively for a specific donor or group of donor countries. Possible examples where the GEF-funded projects may generate exclusive benefits are activities that establish ecotourism facilities in recipient countries with strong traditional ties to a specific donor country. This country may therefore derive extra benefits from the GEF support. Likewise, donor countries with a mature biotechnology sector may benefit more significantly from projects on the conservation of genetic diversity than other countries that have not been able to establish this sector to a similar extent. Overall, the extent to which the benefits in these examples display a substantial exclusiveness in their use needs to be clarified.

When assuming for a moment that certain GEF projects indeed yield exclusive benefits, it may be argued that such projects should be supported in particular, since, according to the theory, this would increase the incentive to contribute and, thereby, limit the extent of free riding. However, while the theory suggests that an increasing proportion of the private good relative to the public good reduces free riding, this does not necessarily imply that more financial resources are allocated to the provision of the public good in total. Moreover, the total supply of the public good either decreases or increases depending upon whether the private and public goods are (Hicksian) complements or substitutes (Cornes and Sandler 1984, 1994). Therefore, a change of this kind in the GEF policy may not necessarily induce increasing funds for global environmental protection.

Multiple Global Public Goods

Suppose that a group of countries does not share only a single international public good but also multiple public goods. Accordingly, the allocation of one good influences the allocation of the other ones and vice versa. Each country not

[236] It turns out that if the ratio of exclusive benefits to total benefits increases, the Nash outcome approaches the Pareto-efficient outcome (Sandler and Hartley 2001).

only has to decide how much to allocate to public goods as a whole, but also how much to allocate to each of these goods given the decisions of the other countries. In this context, the literature studies the extent to which countries become specialized in the provision of specific public goods when assuming their prices, i.e., the domestic costs of providing multiple public goods differ between countries (Boyer 1988, 1990; Kiesling 1974).[237]

Against this background, the figures on the GEF contributions of certain countries that appear to be modest relative to the country's ability to pay represented by the national income do not necessarily serve as evidence of free riding. Moreover, a cost-effective international division of responsibilities could imply that countries make substantial contributions to the provision of other international public goods.[238] In effect, these reflections place the finance of the GEF in a broader context in which several international public goods compete for funding and the community of countries has to assign priorities to various public goods and allocate responsibilities for their financing. A meaningful empirical analysis of this issue is particularly complicated by the fact that it is difficult to identify a systematic relationship between the finance of environmental protection and other international public goods, such as collaboration in military deterrence and peacekeeping or the fight against infectious diseases. Accordingly, the analysis of this issue is outside the scope of my study.

4.4.2.2 Financing of Environmental Protection in Repeated Interactions

The static collective action framework displays great flexibility in incorporating different settings with donor countries showing different characteristics with respect to national income, prices, or behavior. Otherwise, one weakness of the approach is that it considers the donor interactions a one-shot game and, thereby, only describes intertemporal incentive problems in a reduced form.

More precisely, when applied to the finance of the GEF, financing arrangements are modeled as a sequence of finite one-shot games where the payoff space is somehow (exogenously) restricted by the described burden sharing framework. Furthermore, it is implicitly assumed that, for each donor, the actual

[237] When considering a two-country model with two public goods that are produced with constant-returns-to-scale technology and with countries that make constant expenditures on public goods, international coordination leads to an allocation where each country becomes completely specialized in the production of the public good it has an comparative advantage in. As compared to the situation without coordination, in the case of cooperation and shared responsibilities, the utility levels for both countries, as well as the total supply of both public goods, increase (Boyer 1988).

[238] Note that a prerequisite for efficient allocation in this regard is that there are no third countries that receive benefits from the provided public goods without making any contributions (Jack 1991).

contribution coincides with the contribution committed. No donor behaves opportunistically and, thus, no donor needs to adopt a trigger strategy, i.e., if one country contributes less than it has committed, it is profitable for the other donor to react accordingly in the subsequent periods. In other words, the outcome in each (GEF) period is assumed to be wholly unrelated to that of preceding periods.

While the described collective action models place the emphasis upon the agreement on the *size of the trust fund* in a specific GEF period, let us suppose now that donors have agreed to maintain a fund that provides sufficient project funding over some time. Accordingly, the subject of negotiation is a *burden-sharing scheme* that can be sustained in the upcoming GEF periods.

In this conceptual setting, it is possible that some donors deviate from the individual commitment that is derived from the burden-sharing scheme, i.e., some donors free ride. Given the corresponding reactions of the other donors (trigger strategy), the finance of environmental protection as a global public good is affected or even collapses completely. Since donor countries anticipate this incentive problem and the potential adverse impacts connected with it, compatible solutions have to be found and incorporated into an agreement on the burden-sharing framework.

A seminal paper by Barrett (1994a) theoretically analyzes *repeated interactions* of this kind.[239] The paper investigates an agreement on the financing of a global environmental fund whose financial resources are used to support biodiversity projects in host countries of the developing world.

The paper, like many other papers on international environmental agreements (IEAs), relies upon the approach of noncooperative game theory (Barrett 1998; Carraro and Siniscalco 1998; Finus 2004). With these tools, two issues of IEAs are typically investigated: their formation as an economic coalition and their stabilization. Both of these issues refer to the two types of free riding, i.e., the incentive not to participate in an IEA and the incentive to deviate from its agreed terms or withdraw.

In this regard, Barrett (1994a) does not emphasize the formation process of an agreement on the finance of an environmental fund but analyzes primarily the effectiveness of the agreement given the incentive to free ride for a donor country. The key issues are the design of the agreement and the country's decision to comply with the agreement's provisions.[240] The author argues that, by anticipat-

239 The model represents an application of a seminal paper on the self-enforcement of international agreements (Barrett 1994b). This paper provides a more detailed description of the structure and intuition of the model discussed here.

240 For a study of the formation of IEA, see, for example, Carraro and Siniscalco (1998).

ing the free-riding problem, the formation of a coalition only succeeds if a well-designed agreement properly addresses the incentive constraints.

A Dynamic Game Model

In the model, n identical developed countries act as donors where each country decides upon the size of a money transfer, M_i, that it provides to the environmental fund. The fund invests the money in a cost-effective way to enable environmental protection, S, in excess of the level that recipient countries in the developing world consider domestically optimal and already implement in the absence of international assistance.[241]

In the *noncooperative baseline*, each developed country considers the contributions by the others countries to be given (Nash–Cournot behavior). The net benefit, NB_i, of country i is the gross benefit for $B_i(S)$ given levels of protection, S, minus the transfer, M_i :

$$(4.43)\quad NB_i = B_i(S) - M_i \quad \text{with } \frac{\partial B_i(S)}{\partial S} \geq 0, \quad \frac{\partial^2 B_i(S)}{\partial S^2} < 0 .$$

The sum of the transfers, M, equals the cost of total protection $C(S)$, i.e.,

$$(4.44)\quad M = \sum_i M_i = M_i + M_{-i} = C(S) \quad \text{with } \frac{\partial C(S)}{\partial S} > 0, \quad \frac{\partial^2 C(S)}{\partial S^2} > 0 .$$

By solving for S in equation (4.44) and substituting S in equation (4.43), the individual net benefit, NB_i, is described as the function of the individual contribution, M_i, and the contributions by the others, M_{-i}, i.e.,

$$(4.45)\quad NB_i(M_i, M_{-i}) = B_i(M_i, M_{-i}) - M_i .$$

The first-order condition for a net benefit maximum serves as the description of the individual reactions paths, $M_i^*(M_{-i})$, that are, in turn, used to derive a Nash equilibrium. This static part of the model resembles the Nash–Cournot model of private public good provision described in the previous section. Only the national income is not constrained in this representation and the elasticity of substitution between conservation and consumption may differ from the Cobb–Douglas case, depending upon how the benefit function is specified.

Because of the unconstrained income, Barrett (1994a) employs a benefit function, $B_i(S) = (b/n)(aS - S^2/2)$, that yields a linearly decreasing marginal

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[241] The author assumes that developing countries as transfer recipients behave as price-takers, i.e., they do not have bargaining power and are not able to act strategically. In contrast, see Cervigni (1998).

benefit. The cost function is $CS = (cS^2)/2$. The transfer amount, M^*, and the level of conservation, S^*, characterize the corresponding Nash equilibrium:

$$(4.46)\quad M^* = \frac{a^2b^2c}{2(cn+b)^2} \quad \text{and} \quad S^* = \frac{ab}{cn+b}.$$

In the *cooperative outcome*, donors collaborate and form a coalition where each coalition member provides money in excess of the amount in the noncooperative baseline. More precisely, the members of the coalition choose their individual contributions to maximize the coalition's aggregate payoff. The author shows that when all countries join the coalition, the equilibrium terms are

$$(4.47)\quad M_c^* = \frac{a^2b^2c}{2(c+b)^2} \quad \text{and} \quad S_c^* = \frac{ab}{c+b}.$$

However, not all countries may want to be a member of the coalition right away. As a nonmember, a country observes the coalition's decision and determines its contribution noncooperatively in order to maximize its payoff. Alternatively, if an additional country joins the coalition, each of the previous coalition members increases its contributions. In the same way, individual contributions are collectively reduced if a member country withdraws. In other words, the members of the coalition recognize how their decisions influence the actions of the nonmembers (Barrett 1994b).

In order to represent the *dynamic structure* of the game, the author assumes that countries decide on the optimal size of their contributions in the initial stage and carry out the payments in all subsequent periods. To deter any member country of the coalition from free riding, i.e., from providing less than the agreed upon amount in any of the subsequent periods, a *credible threat* has to be imposed: if a country does not comply with the agreed upon terms, it is punished in that the other countries also reduce their contributions in the subsequent periods (trigger strategy) and less environmental protection is safeguarded (Finus and Rundhagen 2001).

Credibility requires the agreement reached to be renegotiation-proof, i.e., member countries do not want to renegotiate after the defection and punishment has occurred, although punishing the defecting countries hurts the inflicting countries.[242] In the model by Barrett (1994a), (weak) renegotiation-proofness requires that, first, no member country has an incentive to free ride when facing

[242] Renegotiation-proofness is a solution concept for repeated games to describe stable equilibria (Farrell and Maskin 1989). There are several nuances in stability concepts that are related to several criteria that have to be fulfilled (Pearce 1992; Bergin and MacLeod 1993).

the conditional punishment, and second, the member countries that inflict the punishment have no incentive to renegotiate the agreement afterwards.

An agreement in this dynamic setting is only stable, or, synonymously, *self-enforcing*, if it satisfies the conditions of renegotiation-proofness.[243] Given these conditions, the author studies the features of the cooperative outcome. The major questions are (1) whether a stable agreement that supports cooperation in an infinitely repeated game exists and, (2) if so, how many countries are signatories to such an agreement. Since it is assumed that there are no barriers to accede to the arrangement, nonsignatory countries apparently believe that they are better off by remaining outsiders. In this regard, the number of signatories to a stable agreement can be explained by the equilibrium concept.

Numerical Simulations and Derived Implications

Barrett (1994a) applies the derived equilibrium terms in numerical simulations with various benefit-to-cost parameters in order to describe the equilibrium number of signatories of a stable agreement. Each parameter combination is transformed into a unique outcome with a specific number of signatories.

It turns out that a full cooperative outcome with a comparatively large number of signatories can only be sustained when benefits and costs of marginal protection behave in such a manner that each cooperative donor in effect only experiences a minor increase in net benefits relative to the noncooperative outcome. For a substantial difference in the individual benefit between the two outcomes, the equilibrium number of signatories is rather small. This implies that a stable agreement with a large number of parties and, hence, with a substantial impact on the financing of environmental protection is difficult to establish.[244]

Against the background of the theoretical results, the author assesses the effectiveness of the Convention on Biological Diversity (*CBD*) as the predominant IEA in biodiversity policy. Since nearly all countries have signed the CBD, the author deduces that this agreement is not likely to lead to a significant improvement in global well-being compared to a situation where it would not

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243 A stable agreement satisfies the following requirements: *profitability*, i.e., it is beneficial for a donor country to join the agreement and provide the agreed contributions to the fund; *participation*, i.e., no country that participates has an incentive to leave the agreement; and *compliance*, i.e., no participating countries will deviate from the terms of the agreement (Finus 2004).

244 Barrett (1994a) points at the two driving forces in the model: the smaller the benefit from protecting a marginal unit, the smaller the negative impact due to non-compliance and, therefore, the smaller the preassigned punishment. On the other hand, large costs of marginal protection induce potential gains from free riding and, thus, increase the incentive for defection. A stable agreement with a large number of signatories ideally requires low absolute costs of protection at a relatively high benefit-to-cost ratio, which usually does not occur in practice.

have been reached. According to the author, the more general implication thereof for environmental protection is that the financial resources the donors provide cooperatively in a multilateral institution are typically insufficient in size. Therefore, it is crucial whether unilaterally provided funds are substantial in order to assist environmental protection and, thereby, fill the gap that exists due to modest financing in a cooperative framework.

In reference to the finance of the GEF, this conclusion needs to be considered in more detail: Barrett (1994a) relates the model directly to the CBD and not specifically to the GEF as the actual "global environmental fund" that is associated with the CBD. In this regard, the comparison of theoretical results and empirical findings is based upon a slight misuse of the terms: the model assumes that only "developed countries" provide money for transfers, which roughly holds for the GEF. However, in his conclusion, the author refers to the CBD explicitly, which is signed by developed and developing countries (178 countries). It would be more precise to compare the actual group of GEF donors (previously 35 countries at maximum) with the group of countries that are assumed to receive (substantial) external benefits from environmental protection in the developing world and that are therefore expected to contribute to the fund.[245]

Furthermore, to describe the cooperative outcome, the model distinguishes donors who cooperate in order to finance the environmental fund from donors who contribute to the same fund on a noncooperative basis. In the author's conclusion, the latter are referred to as unilateral transfers. In this respect, the "fund" apparently represents a conceptual vehicle to describe the north-south transfer in a simplified way.[246] When using the theoretical model to explain the finance of the GEF, contributions by the members of the subgroup in the cooperative outcome at best represent the GEF as the actual multilateral fund.

The actual burden-sharing arrangement in the replenishment process may be interpreted as a stable international agreement that is Pareto efficient for the group of signatory countries. Suboptimality only results from the fact that not all

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245 The size of the external benefits a country receives from the GEF investment can hardly be identified. Depending on whether it is believed that only wealthy developed countries receive external benefits and that therefore they should be the only donors to the GEF, the deductive conclusion that there are relatively small gains from multilateral cooperation may be preserved, since the donor community is actually represented by the G7 in the first place. Otherwise, there is reason to believe that countries that have been outside the donor community, also receive external benefits. When in this context, only about 30 countries out of nearly 180 are donors to the GEF, this may suggest that the other extreme of the theoretical findings applies, namely that in a small group of donors, each of them experiences a substantial increase in well-being from cooperation.

246 Note that in this respect, it is assumed that unilateral donors still have full bargaining power relative to developing countries as transfer recipients.

countries are signatories, i.e., participate as donors (Finus 2004). In order to increase the total contributions to the GEF, it might be conceivable to replace the existing arrangement with another stable agreement that is connected to a larger number of donors, whereas all of the previous donors contributes at least as much as before.

When applying the theoretical model, this reasoning implies that there are multiple equilibria with regard to the number of signatories and it is possible to move from the existing equilibrium to the equilibrium with more signatories and the largest possible size of the GEF fund. Nevertheless, the model does not provide evidence that multiple equilibria indeed exist. In fact, if only one unique equilibrium were attainable, this would imply that it is not possible to increase the size of the GEF fund in this way. Following the model by Barrett (1994a), the resulting size of the fund may change only if parameters that are exogenous to the model change.[247]

Repeated Interactions and Asymmetric Countries

In the Barrett model, the question of cooperation and the number of signatory countries is discussed against the background of identical donors. In comparison to the static collective action model, this dynamic model neglects asymmetries between donor countries.[248]

In practice, *asymmetries* may, on the one hand, refer to the extent to which a country's well-being is affected by an undersupply of global environmental protection. For example, if the benefit from environmental protection per capita is identical, a highly populated country benefits to a larger extent than a less populated country. Similarly, geographical characteristics of a country can determine the degree of vulnerability to the impact of the depletion of the ozone layer or a sea level rise, which is induced by climate change and, therefore, influences the benefit received from environmental protection (Boadway and Hayashi 1999; Sandler and Murdoch 1997). On the other hand, asymmetries may refer to national income and the economic capacities of a country, i.e., the ability to contribute to the GEF. Both forms of asymmetry can be interrelated.

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[247] For example, it is unlikely that the size of the GEF fund will increase if the costs of environmental protection in developing countries rise or if the benefit the donor countries receive decreases.

[248] Furthermore, there is a set of advanced techniques in dynamic noncooperative game theory that have recently been applied to IEA. In this regard, Finus and Rundhagen (2001) critically mention the model's feature that implies that only one coalition can be formed at any one time.

The Impact of Income Inequality

Itaya and Yamada (2004) consider the influence of asymmetry in national income and analyze whether a self-enforcing agreement between asymmetric donor countries can be maintained or not. Their model considers an infinitely repeated contribution game with two asymmetric countries where preassigned punishments are applied to sustain a Pareto-efficient allocation as a weak renegotiation-proof equilibrium. The model initially corresponds to the static model of Lindahl equilibrium presented in the previous section. It is extended to a dynamic game by assuming that the two countries repeat the contribution game over an infinite horizon.

In each period, there is an incentive for each country to contribute less than the amount necessary to implement the Lindahl outcome. If one country defects, the other conducts the punishment by also reducing its contribution (trigger strategy). The defecting country then has to decide whether to return to the Pareto-efficient contribution in the next period (repentance) or continue with noncooperation (retaliation). Expected future payoffs are discounted by an exogenous discount factor, $\delta \in [0,1)$.

By anticipating the possible interactions and the associated payoffs, the Lindahl equilibrium can only be sustained if the agreement between the two countries that establishes the equilibrium is self-enforcing, meaning that it satisfies the conditions of weak renegotiation-proofness (Itaya and Yamada 2004).[249] In order to study whether the conditions are technically satisfied, these conditions are transformed to define a critical discount factor, $\delta^{\min}$, which is compared with an exogenous actual discount factor. A self-enforcing agreement requires the actual discount factor to exceed the critical threshold for both countries, $\delta > \delta^{\min}$. The actual discount factor is assumed to be identical for both countries, while the critical discount factor varies between them, since it depends upon the relative income level. The authors investigate how the extent of income inequality interacts with the stability requirements.

For Cobb–Douglas preferences and a one-period punishment, it turns out that cooperation is more likely to occur, the smaller the differences between income levels are, or, in other words, income inequality makes cooperation more dif-

[249] The authors name the four conditions of this solution concept. Since one of them is redundant and another is satisfied trivially, i.e., does not impose an additional restriction, the equilibrium conditions are reduced to the following two: first, the discounted sum of utilities in the case of defection in one period is smaller than that in the case of cooperation in all periods and, second, the discounted sum of utilities from investing in a return to cooperation after punishment has been inflicted is greater than the utility from continuing with noncooperation (Itaya and Yamada 2002).

ficult to attain.[250] Furthermore, it is shown that the incentive to deviate from the agreed upon terms is greater for the low income country.[251]

Regarding the funding in the finance of the GEF, it is shown that countries with the highest ranks in national income are indeed donors to the GEF, in particular the G7 countries, or, more broadly defined, the OECD DAC countries. To repeat, a way to increase the total contributions to the GEF would be to convince transition countries and newly industrialized countries to accede future replenishment agreements. The theoretical finding, in turn, implies that it is quite difficult to reach a stable arrangement for a larger donor community whenever the potential new donor countries have a lower national income (in absolute terms) than most of the existing donor countries. In this regard, it may be difficult to increase the total size of the GEF fund by expanding the donor community.

Side Payments

When asymmetries between countries impede cooperation between them, a way to assist cooperation is to influence the profitability constraint of the noncooperating countries. In this context, the literature discusses the role of transfers, or, synonymously, side payments, that signatory countries provide to nonsignatory countries (Carraro and Siniscalco 2001; Siebert 2005: 214f.). Side payments are *incentive payments* that serve to make a country undertake a commitment to international cooperation it would otherwise be unwilling to do. These payments represent money transfers and/or in-kind transfers (Barrett 2003: 336).

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250 In two subscenarios, the authors also consider CES preferences and quasi-linear preferences. While in the latter case, income inequality does not have an impact upon the results, numerical simulations for the CES case generally confirmed the findings of the Cobb–Douglas case (at least for a wide range of parameter values). In addition, the authors discuss the impact if the punishment is expanded to more than one period. It is concluded that it would become even more difficult to obtain cooperation (Itaya and Yamada 2002).

251 The authors remark that this result is reminiscent of the findings by Olson (1965) on the "exploitation hypothesis," which states that when providing an international public good in collective action, the wealthier country carries a *disproportionally* large burden of the supply costs. While this result was originally derived in a static setting, it is actually reproduced in a dynamic setting in Itaya and Yamada's model (2002).

The reference to Olson's findings also provides evidence of the economic logic behind the theoretical results. According to Olson and Zeckhauser (1966), the reason why wealthier countries indeed bear a disproportional large share of the costs is that they place a higher value upon the collective good than the less wealthy countries and, therefore, have fewer incentives to free ride. In the dynamic model, this is expressed in the relatively lower critical discount factor of the high income country.

The major question addressed in the literature is whether side payments induce cooperation or the enlargement of an existing coalition in a self-enforcing agreement (Barrett 2003: 335ff.; Carraro and Sinscalco 2001; Chandler and Tulkens 1997). Based upon their theoretical modeling, it turns out that side payments only have a stabilizing impact upon cooperation under specific conditions, such as a certain form of commitment on the part of the member countries (Carraro and Sinscalco 1993) or strong asymmetry (Barrett 2001).[252]

When applying the idea of side payments to the *finance of the GEF* and the use of GEF funding to safeguard environmental resources in the developing world, it is helpful to distinguish three groups of countries. First, countries that receive considerable benefits from resource protection contribute to the GEF. Second, countries that only derive low benefits from protection have not joined the financing of the GEF. Finally, (developing) countries that receive transfers from the GEF enforce measures on environmental protection within their territory.[253] When actions in the developing countries are related to provisions of IEAs, the GEF transfers to countries in the third group are in effect side payments of the first group of countries in order to increase participation in a specific IEA (Barrett 2003: 350). Considering the relationship between the first and second groups, the question is whether there is also a side payment arrangement in order to reinforce cooperation and increase the total contribution to the GEF fund.

Such a specific interaction between the three groups of countries is not addressed in the literature: Barrett (2001) and Carraro and Sinscalco (1993) only consider two types of countries. As described above, Barrett (1994a) implicitly assumes three groups of countries, whereby recipient countries always participate, i.e., they enforce conservation whenever they receive sufficient transfers from the other countries and there are no side payments among the first and second groups. Yet, countries in the second group provide some funding on a noncooperative basis.

An intuitive answer to the question on side payments has to take into consideration the *incentive constraints* that side payments have to satisfy (Carraro and Siniscalco 1993). First, it must be profitable for the recipient of side payments to accede to the arrangement and comply with the obligation to contribute, i.e., the

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252 Both the models by Barrett (2001) and Carraro and Sinscalco (1993) represent one-shot games. The former model considers asymmetry with respect to the benefit received from an abated (global) public "bad," while the contribution to the abatement of each country is fixed at an identical level. Differences in national income are not considered. In equilibrium, a subgroup of countries abates, while the rest does not.

253 For simplicity, suppose that the benefit these countries derive from protection in the other recipient countries is negligible.

enlarged coalition of donors must be self-enforcing. At the same time, side payments must be self-financing, i.e., the contributions by newly acceded donors must generate an additional benefit to the (cooperative) donors that exceeds the value of the side payments.

Since contributions to the GEF trust fund by new potential donors have to be made in money, it is easy to see that side payments by the previous donors only satisfy the incentive constraints if they are made on an noncash basis. An example in this regard is in-kind payments in terms of goods or services that are produced at comparatively low cost in the donor countries and transferred to the new donors in return for their accession to the financing agreement.

In practice, official documents do not provide evidence of in-kind payments that are declared to support future replenishments. Otherwise, undeclared side payments may exist but are difficult to detect. Regarding the second and third GEF replenishment, only two countries, namely Slovenia and Nigeria, joined the coalition as new GEF donors, while several countries withdrew as donors after the first or second replenishment (see Section 4.4.1). It can therefore be supposed that side payments have previously not been considered a means to increase the funds for the GEF or that they have not created a remarkable impact upon attracting new donors.

4.4.3 Burden Sharing in the Finance of the GEF: An Empirical Analysis

Regarding the supposed shortfall of provided resources, it would be interesting to know whether it can indeed be attributed to free riding and which countries could be considered free riders, i.e., countries that do not contribute while receiving external benefits, or "cheap riders," i.e., countries that contribute something, but less than the benefits they are assumed to receive.

In the literature, this question is investigated in the context of (interstate or international) public goods other than environmental protection (Dudley 1979; Khanna 1993; McGuire and Groth 1985; Sandler and Murdoch 1990). By assuming a theoretical framework for the private provision of a pure public good (Cornes and Sandler 1985, 1996: 143ff.), the studies make use of econometric methods to test the alternative behavioral assumptions in collective actions empirically.[254]

[254] Studies by Khanna (1993), Sandler and Murdoch (1990), and McGuire and Groth (1985) suggest estimating individual demand functions for the pure public good under alternative behavioral assumptions (Nash–Cournot versus Lindahl) to qualify which of the assumptions more likely applies to an observed set of contributions. Given the empirical results, observed Lindahl behavior implies no, or less, free riding than Nash–Cournot behavior.

When considering the size of the GEF trust fund as a proxy for the global public good "environmental protection," the econometric methods presented in the literature may be applied to the finance of the GEF in order to identify potential free-riding behavior. However, the data set, especially the time series, of the relevant variables is too small, so it cannot be expected that the econometric approach will yield reasonable results.

To study free riding, I proceed differently. I focus on a descriptive analysis of how the financial burden in financing the GEF fund is allocated among the donor countries that are assumed to receive benefits from GEF funded activities in environmental protection including biodiversity. Initially, I assess the results against the background of normative concepts on equitable burden sharing. In a second step, I discuss the empirical findings with respect to their implication for the potential free-riding behavior of certain GEF donors.

For my analysis, I conceptualize the financing of the GEF such that donor countries collectively finance environmental protection in the developing world as a global pure public good (Cornes and Sandler 1996: 143ff.). I suppose that the financial resources the donors make available for this purpose are granted to projects that are enforced in a fairly cost-effective way. Therefore, the total amount of the collectively provided resources, i.e., total contribution to the GEF, approximates the quantity of the global public good generated (Siebert 2005: 59f).[255]

4.4.3.1 Concepts of Burden Sharing and Exploitation

For the study of burden sharing, the two major questions are:

- How is the burden of providing the public good allocated between the countries that receive benefits from its provision?
- Does the observed allocation coincide with normative concepts of equitable burden sharing?

With respect to the normative question, the literature defines two general concepts. Both are derived from the theory of taxation: the first is the benefit principle, the second the ability to pay principle (Musgrave 1993; Sandler 2002).

The *benefit principle* implies that countries should make contributions according to the size of the external benefits they receive from international environmental protection. In other words, countries that receive relatively large

[255] For simplicity, I suppose that any joint products from environmental protection that only provide domestic benefits are completely financed by the host countries. In other words, incremental domestic benefits from environmental protection in excess of the domestically optimal level are subtracted from compensation payments for incremental conservation (see Section 4.3).

benefits should contribute more than others do. The benefit principle corresponds to the determination of Lindahl prices, i.e., the Lindahl process for efficient provision of the public good simultaneously establishes a burden-sharing rule. Equity in this regard refers to the financing and consumption of the collective good only and has to be separated from equity considerations with respect to income distribution.

The *ability to pay principle*—as far as it is applied in tax policy—is often considered the most equitable form of determining contributions to the provision of public goods. In contrast to the benefit principle, burden sharing is decoupled from the benefits received; it is based upon considerations of fairness and distributive justice. The principle implies that countries with identical income should contribute the same amount (horizontal equity), while countries with a larger income and, therefore, a greater ability to pay should contribute a larger amount (vertical equity). The issue in this context is how to determine appropriate differentials in contributions for countries with different income levels. In particular, the question is whether contributions should increase on a progressive scale relative to the national income (Musgrave 1993).[256] Differences in the contributions derived from each of the two concepts are reduced if countries have fairly similar preferences and the public good displays significant positive income elasticity, i.e., the larger the national income, the larger the benefit the country derives from it.

Regarding the finance of the GEF, the official documents report that the contributing participants have agreed upon a fair burden-sharing arrangement that is, inter alia, guided by the ability to pay principle (GEF 2001a).

The Exploitation Hypothesis

Several studies on collective action theory investigate burden sharing for providing international public goods. These studies typically assume Nash–Cournot behavior for the donor countries involved. A major finding in this regard is summarized in the *exploitation hypothesis* (Olson and Zeckhauser 1966; Sandler 1992: 54ff.).[257]

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[256] To determine appropriate differentials in contributions, the principles of equal sacrifices can serve as underlying concepts. However, not all of these principles necessarily imply a relationship of a progressive scale (Musgrave 1993).

[257] Exploitation means that the surplus generated from collective actions is distributed in such a way that some countries (the exploited ones) systematically receive a share of the generated surplus that is disproportionately smaller than their contributions would justify. At the same time, the other countries manage to obtain a correspondingly larger share. Exploitation, in this respect, refers to overall utility levels. In fact, the standard model of voluntary public good provision as presented in Andreoni (1988) or Bergström et al. (1986) implies that in equilibrium, every contributing country

In their seminal paper, Olson and Zeckhauser (1966) use a static two-country model of the voluntary provision of a pure public good. The authors assume that the marginal cost of provision is constant and that countries have identical preferences but are of a different economic size, i.e., they demonstrate *different levels of national income*. It is then shown that in the Nash equilibrium, in comparison to any Pareto-efficient outcome, the total supply of the public good falls short of the Pareto-efficient level, and that the large countries, or, synonymously, the high income countries, carry a disproportionately large share of the supply burden. In contrast, the low income countries carry a burden that is less than their share of the surplus generated from collective action. Thus, the *low income countries* exploit the high income country by behaving as *cheap riders* (Kwon 1998).[258]

This theoretical finding is based upon the assumptions that each donor country acts as single agent and that each national government acts on behalf of the representative citizen. Thus, differences between donor countries with respect to *population size*, or *income per capita*, are disregarded and not captured in the analysis. When other things are held equal, but population size differs, countries with a large population may free ride less than countries with a small population. This is because, since the benefits of international pure public goods accrue to each citizen, large countries internalize a comparatively greater share of their own contributions. Consequently, it is reasonable for a high income and largely populated country to make a disproportionately large contribution.

Nevertheless, in the same respect, a small country with an exogenously high income per capita may only contribute a comparatively small amount. Its national income in absolute terms may yet be greater than the income of a largely populated country with a comparatively low income per capita. If the latter low income country makes a comparatively larger contribution, the finding of an exploitation of the high income country is not confirmed (Boadway and Hayashi 1999). This simple example implies that when assessing the burden sharing in

(with identical preferences) attains the same utility level regardless of the level of national income.

[258] The economic intuition for this result is the following: suppose that by definition, the high income countries place a relatively higher absolute value upon the collective good (Olson and Zeckhauser 1966). Then, each country makes contributions to the provision of good in order to satisfy its own demand. To determine the size of the contribution, the marginal cost of supply is equated with the individual marginal benefit. Since external benefits for the other countries are not taken into account and the individual marginal benefit is decreasing, each country has an incentive to stop making contributions before the Pareto-optimal output of the good is attained. Since, by definition, low income countries derive a lower value from the good, they have a comparatively lower incentive to contribute and stop investing before the high income countries. As a result, there is disproportionate burden sharing (Sandler 1992: 54ff.; Olson and Zeckhauser 1966).

financing an international public good, the influence of differences between population and income per capita has to be taken into account.

Since it is generally agreed that the setting underlying the outlined model applies to the provision of many international public goods, the theoretical outcome of an exploitation of the high income countries is *empirically tested* for several of them. The focus in this regard falls upon the funding of military deterrence by the NATO alliance (Olson and Zeckhauser 1966; Sandler and Forbes 1980; Sandler and Murdoch 2000). In addition, the financing of development assistance or the finance of United Nations institutions is frequently analyzed (Addison et al. 2004; Kwon 1998; Olson and Zeckhauser 1966).

Both parametric and nonparametric tests are applied. The use of *nonparametric tests* is justified because of data limitations and the nature of the data, i.e., when the set of available observations is so small they cannot be described by parametric distributions (Olson and Zeckhauser 1966; Sandler and Forbes 1980). In order to investigate disproportionality, the studies relate absolute or relative measures of the individual contributions to the size of the national income.

The results of the empirical testing vary across the studies.[259] Addison et al. (2004) and Kwon (1998) find evidence of reverse exploitation in the financing of development assistance, i.e., donors that are comparatively small in economic size carry a disproportionately large share of the total burden. This is worth mentioning, since the GEF contributions are related to the burden-sharing scheme of the IDA, which is a multilateral institution of development assistance (see Section 4.4.1).

4.4.3.2 Analysis of Horizontal and Vertical Equity

Given the different principles of burden sharing and the requirement that the ability to pay principle should guide the finance of GEF, I study (1) the extent to which the ability to pay among GEF donor countries explains the burden sharing observed and (2) whether there is indeed exploitation within the community of donors.

I begin with an analysis of whether horizontal and vertical equity in the simple definition were satisfied in the previous GEF replenishment processes. While this analysis is based upon the absolute size of GEF contributions as a measure of burden, I subsequently consider relative burden measures to describe the impact of differentials in contributions and ability to pay. Due to the small data set, I use nonparametric methods.

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[259] Several papers provide an overview of the empirical literature, for instance, Addison et al. (2004), Kwon (1998), and Sandler and Hartley (2001).

Horizontal and Vertical Equity

Following the definition of horizontal and vertical equity, burden sharing that is based upon the ability to pay principle should meet two conditions. First, donor countries with a national income of identical size should contribute the same amount to the GEF trust fund (horizontal equity). Second, for any pair of donor countries, the one with the higher income should make a larger contribution (vertical equity)—or, at least, a contribution that is not smaller than that of any other country with a lower income. I name this condition weak vertical equity.

I define the measure of *burden* as the contribution, in absolute terms, that each donor country commits in a replenishment process.[260] In order to express a donor's *ability to pay*, the literature typically uses a definition of national income. I employ figures on annual gross national income (GNI) in current dollars (determined using the Atlas method) and, alternatively, the GNI in international purchasing power parities (PPP).[261]

I study the contributions for each four-year GEF period as the measure of burden, and the donor's GNI calculated over the corresponding period. Next, I determine ranks for each donor country with respect to its contributions and its income and compare these ranks. Since the data demonstrate no two donor countries have a GNI of the same size, the analysis considers *vertical equity* only.

To satisfy the strict form of vertical equity, the donor country's burden ranking must be identical to its ability to pay ranking. However, strong vertical equity is not satisfied as long as the minimum contribution requirement (MCR) is binding for several donor countries, i.e., as long as the contribution exactly equals the MCR for several countries. As described in Section 4.4.1, this is the case with several countries in all three replenishments.

Consequently, I only analyze the weak form of vertical equity. For this purpose, I order countries that display an identical rank in the burden measure, i.e., countries that have contributions of the same size, descendingly according to their GNI and adjust the rank assignment for the burden measure accordingly. Considering the two resulting vectors of ranks, *weak vertical equity* is satisfied if the ranks for the burden and the ability to pay display a significant positive correlation.

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260 In order to describe the contributions, I take data from various official documents (GEF 2002c, 2003, 2004b) (see Section 4.4.1).

261 The information on income levels is taken from the 2005 World Development Indicators database. Figures on GDP for 2004 to 2006 are calculated by using OECD real GDP growth projections. For the few donor countries, for which no growth projection is provided in this source, I assume average growth rates that have been observed for the preceding four-year period.

The empirical data on contributions already illustrates that there is, in fact, no perfect positive correlation between the two measures. Considering the countries that participate in all three replenishments only, several recipient countries, such as China, India, Korea, and Mexico, always show a lower rank for burden sharing than for ability to pay. The same is true of the nonrecipient country Spain. In contrast, for nine nonrecipient countries of medium economic size, as well as the recipient countries Côte d'Ivoire and Pakistan, the rank for burden sharing exceeds that for ability to pay. For the remaining countries, the difference between the ranks is either zero or varies between the replenishments.[262]

I summarize the results of these ranks by calculating *correlation coefficients* for the two rank variables in each GEF period. Commonly used tools for this nonparametric analysis are the Spearman rank correlation coefficient (ρ) and the Kendall rank correlation coefficient (Kendall τ) (Sandler and Murdoch 2000). Given the sample of bivariate observations, I calculate the rank for each element in an observation and compare it with the same element in the other observations. By using the resulting vectors of ranks, I determine a correlation measure with a potential range from -1 (perfect inverse correlation) to $+1$ (perfect correlation).[263]

Table 23 presents the results for both coefficients. Each row in the table describes the correlation for a specific GEF replenishment. It turns out that the size of the contribution is positively correlated with national income in all three replenishment processes.

The numerical results on rank correlation can be used to carry out hypothesis tests on the relationship between contributions and income. It can be shown that a null hypothesis stating that both variables are uncorrelated is rejected at significance levels close to zero. Thus, there is a tendency for the larger values of income and contributions to be paired. I, therefore, conclude that the ability to pay principle, in terms of a weak vertical equity, has been applied in the previous finance of the GEF.

In the previous analysis, I have not considered any relationship between differentials in contributions and differentials in national income. In the literature, this relationship is taken into account by defining the burden in relative terms rather than absolute terms. There are two measures in this regard: either

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262 The classification is based upon figures for gross national income (GNI) in current dollars (Atlas method). The nine nonrecipient countries are Austria, Belgium, Canada, Denmark, Finland, the Netherlands, Norway, Sweden, and Switzerland.

263 While calculating the Spearman rank correlation is less complicated, there is no explicit definition in economic terms. The Kendall τ in turn is defined in terms of the probability of observing concordant and discordant pairs (of ranks). In most cases, the values of the Spearman and the Kendall rank correlation are very close and, therefore, lead to the same conclusion (Conover 1971).

Table 23:
Rank Correlation between Contribution Size and Income

	Observations	Spearman rank correlation coefficient		Kendall rank correlation coefficients	
		ρ_{contr} GNI ($)	ρ_{contr} GNI (PPP$)	τ_{contr} GNI ($)	τ_{contr} GNI (PPP$)
GEF-1	35	0.787	0.638	0.630	0.496
GEF-2	36	0.840	0.693	0.721	0.587
GEF-3	32	0.790	0.647	0.649	0.524

Notes: ρ_{contr} GNI denotes the Spearman ρ of GEF contribution to GNI. τ_{contr} GNI denotes the Kendall τ of GEF contribution to GNI.

the within-ally burden, i.e., a donor's contribution in absolute terms as a percentage share of national income, or the between-ally burden, i.e., the individual contributions as a percentage of the total contributions of the group of donor countries (Sandler and Forbes 1980). For this purpose, I use the measure of the *within-ally burden*, i.e., for country *i*, $b_i = contrib_i / income_i$.

Analysis of Differentials in Contributions and National Income

The question for the nonparametric analysis is whether for a pair of donor countries, *i* and *j*, with $income_i > income_j$, it holds that $b_i > b_j$ rather than that $b_i < b_j$. Expressed in ranks, this is a positive correlation between the rank figures of the two variables. To study the question empirically, I first calculate the values for the burden b_i by using data on the actual donors' contributions in each of the three GEF replenishments and the annual GNI in current dollars (Atlas method).[264] I define the ability to pay as the present value of annual GNI during a corresponding GEF period.

Given the figures for the contributions relative to the GNI and for the GNI itself, I determine the ranks for both variables for each country. Based upon this, I derive the Spearman and Kendall rank correlation coefficient for correlation between the two variables (Conover 1971: 245ff.)

Table 24 displays the *correlation* for each replenishment process. Results are given for the group of *all donors* that participate in a replenishment process and for three subgroups thereof. The second group includes all donors that are

[264] Since GEF contributions refer to a four-year GEF period but income data are given on an annual basis and I have no information on how the actual contributions are allocated across the GEF period, I transform contributions and GDP into the same units: I suppose that donor payments within each GEF period are divided into annual tranches of an identical nominal size. The burden for each GEF period is defined as the present value of these annual payments divided by the present value of annual national income (GDP).

Table 24:
Burden Sharing: Rank Correlation between Within-Ally Burden and Income

	Donor group (observations)	Spearman rank correlation coefficient	Kendall rank correlation coefficient
		$\rho_{b\,\mathrm{GNI}}$	$\tau_{b\,\mathrm{GNI}}$
GEF-1	All (35)	–0.360 (0.03)	–0.298 (0.12)
	Nonrecipient (23)	–0.202 (0.37)	–0.178 (0.25)
	EU (15)	–0.032 (0.91)	–0.010 (1.00)
	excMCR (19)	–0.470 (0.04)	–0.345 (0.04)
GEF-2	All (36)	–0.332 (0.05)	–0.270 (0.02)
	Nonrecipient (23)	–0.253 (0.25)	–0.202 (0.19)
	EU (15)	–0.061 (0.83)	–0.048 (0.84)
	excMCR (19)	–0.597 (0.01)	–0.404 (0.02)
GEF-3	All (32)	–0.224 (0.22)	–0.190 (0.13)
	Nonrecipient (23)	–0.172 (0.43)	–0.146 (0.34)
	EU (15)	–0.079 (0.78)	–0.048 (0.84)
	excMCR (24)	–0.250 (0.24)	–0.174 (0.24)

Notes: b denotes the burden proxy. $\rho_{b\,\mathrm{GNI}}$ denotes the Spearman ρ of the GEF burden to GNI. $\tau_{b\,\mathrm{GNI}}$ denotes the zero-order Kendall τ. The figures in parentheses indicate the probability of a type I error when testing the null hypothesis of independence in a two-tailed test.

nonrecipient countries, i.e., 22 DAC countries plus the Republic of Korea. The third group consists of the European Union (*EU-15*) only. Since it is expected that the MCR has an impact upon burden sharing, I introduce a fourth subgroup that only includes donors whose contributions in a replenishment actually exceed the MCR constraint (*excMCR*).

The figures in the table indicate (1) that for all groups in the different replenishments, the correlation between relative burden and the ability to pay is *rather low* and (2) that the correlation is *generally negative*. This implies that (1) although there does not seem to be a robust relationship between differentials in the donors' ability to pay and differentials in their contributions to the GEF (within-ally burden) (2) the former is related to the latter in a rather disproportional and reverse manner.

I compare the numerical results for the different subgroups within a GEF period and across the different GEF periods. When comparing the correlation in the subgroup of *nonrecipient countries* with that for the group of *all donors*, it turns out that the correlation is still negative but to a relatively smaller extent, and this applies to all previous replenishments. This finding implies that disproportionality is partly driven by members of the all donor group that are not included in the nonrecipient group. These are recipient countries that show some ability to pay, especially as compared to the low income donors in Europe. Countries that come into question in this regard are primarily China and India, as well as Turkey and some Latin American countries, such as Brazil and Mexico.

A comparison of the result for the *nonrecipient group* with that of the *donors of the European Union* reveals that the correlation is less pronounced. Thus, the negative correlation within the nonrecipient group can be attributed to countries like the United States, Japan, Korea, and Australia.[265] Furthermore, the correlation almost vanishes for the group of EC-15 donors; however, France, Germany, Greece, Italy, Spain, and the United Kingdom can be identified as countries that, on average, possess a higher rank for ability to pay than for burden sharing.

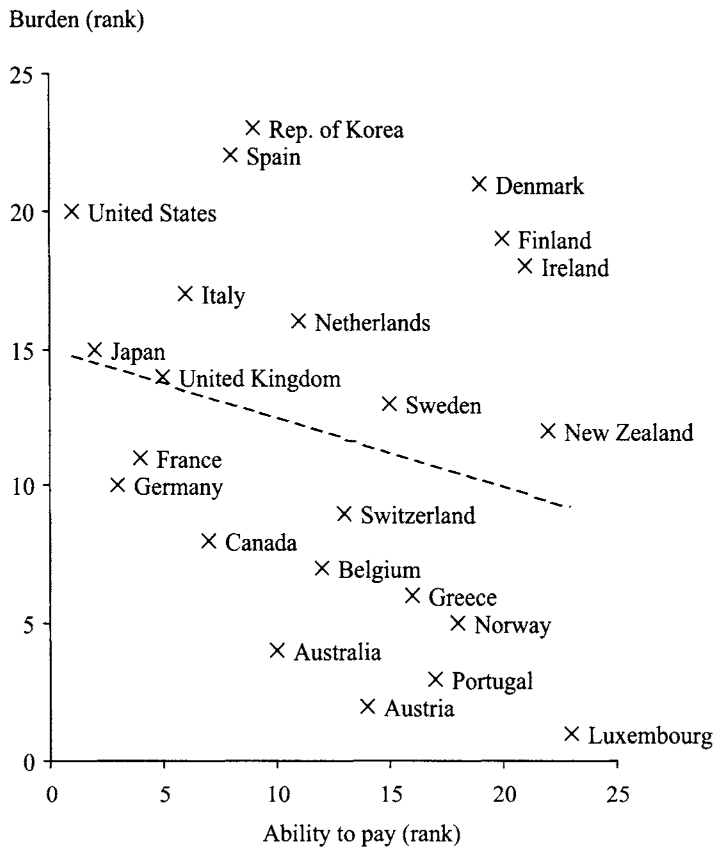
Finally, I compare the correlation for the *all donor* group with the correlation for the *exMCR* group, which only includes countries that contribute in excess of the MCR. It is shown that the negative correlation is even greater for the exMCR group than the all donor group, particularly in the first and second replenishment. This difference implies that the participation of countries whose contribution merely satisfies the MCR, such as Eastern European countries, Côte d'Ivoire, and New Zealand, partly reduces the negative correlation. In other words, these countries carry a disproportionally large burden.

To illustrate, Figure 22 describes the ranks for burden sharing and ability to pay, as well as the correlation between them for the nonrecipient countries in the GEF-2 replenishment. The horizontal axis describes the decreasing rank for ability to pay; the vertical axis indicates the decreasing rank for burden sharing.

If the fairness of burden sharing is defined strictly, i.e., the ability to pay rankings should correspond to the relative burden rankings; all countries have to lie on the diagonal line going from the origin to the upper right corner of the diagram. Considering the actual distribution in the figure, countries located in the lower right corner carry a disproportionately large burden as they have a comparatively low ability to pay (Greece, Luxembourg, Norway, and Portugal). In contrast, countries located in the upper left corner contribute less than they are expected to be able to (the United States, Republic of Korea, Spain, and Italy). However, this interpretation rests upon the underlying definitions of the burden and ability to pay indicators. In accordance with the derived correlation coefficient (–0.253), the dotted trend line in the figure is downward sloping.

[265] To identify the countries that are relevant in this regard, I add the differences between the burden-sharing rank and the ability to pay rank across the replenishments.

Figure 22:
Burden Sharing and Ability to Pay in the GEF-2 Replenishment



Note: A country's burden measure is defined as its rank in GEF-2 contributions as a percentage of GNI. The ability to pay is defined as the country's GNI rank (98–01; Atlas method).

Testing the Exploitation Hypothesis

The disproportionality between income differentials and contribution differentials is reminiscent of the discussion on the exploitation hypothesis. In the literature, empirical testing of this hypothesis usually makes use of ranks for the within-ally burden and national income (Olson and Zeckhauser 1966; Sandler and Forbes 1980). Based upon derived correlation coefficients, confidence intervals are calculated and hypothesis tests are carried out. To confirm exploitation of either the high income or the low income donors, a statistically significant correlation coefficient has to be derived (Olson and Zeckhauser 1966; Kwon 1997).

Considering the finance of the GEF, I use the derived critical values described in parentheses in Table 24, which represent the smallest level of significance at which the null hypothesis of independence is rejected. I discover that a significant relationship does not prevail for the subgroups of nonrecipient countries and EU-15 countries. In this respect, exploitation cannot be confirmed. In contrast, for the all donor group and the group of donors for whom the MCR is not binding, a significant relationship is indicated for both the first and second replenishment. Since the correlation coefficient is negative, it can be concluded that there is exploitation of the low income countries (*reverse exploitation*) in these subgroups.

Ability to Pay and Differences in Population Size and Income Per Capita

Since differentials in national income among donor countries can result from differences in income per capita and population size, I discuss how these differences influence the empirical findings. More specifically, the derived disproportionality may result from the fact that wealthy but less populated countries make small contributions relative to their income and/or that poor but highly populated countries make comparatively high contributions. Following Boadway and Hayashi (1999), the behavior of both types of countries is rational and driven by the extent to which their own contribution is internalized within their own population.

If the difference in population size indeed dominates the findings on burden sharing, the disproportionality indicated should be compensated for or, at least, reduced when correcting the correlation results for the impact of this difference. The methodological tool for such a correction is the *Kendall partial rank correlation coefficient*, which describes the correlation between two variables when the impact of a third variable is held constant (Conover 1971: 253f.; Olsen and Zeckhauser 1966: Footnote 26).

Given the ranks for the within-ally burden and the national income, I also calculate ranks for the *population size* of the donor countries. In Table 25, the partial rank correlation is determined for the group of *all donors* only. The results of the correlation serve as a benchmark and are indicated in the second column. The right-hand columns describe the correlation between the within-ally burden and the ability to pay when the impact of differences in population size is held constant.

A comparison of these correlation results shows that the sign of the correlation does not change although the extent of negative correlation is reduced. This

Table 25:
Rank Correlation between Within-Ally Burden and Income with the Impact of Population Held Constant

All donor group (observations)		Kendall rank correlation coefficient $\tau_{b\,\mathrm{GNI}}$	Kendall partial rank correlation coefficient $\tau_{b\,\mathrm{GNI.population}}$
GEF-1	(35)	–0.298	–0.132
GEF-2	(36)	–0.270	–0.100
GEF-3	(32)	–0.190	–0.032

Notes: b denotes the burden proxy. $\tau_{b\,\mathrm{GNI}}$ denotes the zero-order Kendall τ. $\tau_{b\,\mathrm{GNI.population}}$ denotes the Kendall τ of GEF burden to GNI with the population held constant.

implies that differences in population size, which are included in the uncorrected coefficients, have a reinforcing impact upon the overall reverse relationship between differentials in income and relative contributions. Consequently, the corrected correlation supports the hypothesis that populated donor countries put more effort into the financing of global environmental protection than less populated countries. Nevertheless, this impact is not overwhelming, since anecdotal evidence from figures on ranks for burden sharing indicates that many less populated countries, such as Finland, Norway, and Switzerland, still carry a larger relative burden than densely populated donor countries like China, India, and Turkey.

4.4.3.3 The Interplay between Equity and Efficiency: A Discussion

The empirical findings presented so far are derived against the background of equity considerations. It turns out that when considering the set of actual contributions in the previous GEF replenishments, the normative principle of vertical equity as an integral part of the ability to pay principle is indeed satisfied. Otherwise, there are no explicit arrangements for how to treat relative differences in contributions to the GEF when donors show differences in ability to pay (national income). The preceding analysis indicates that the interactions between donor countries in refinancing the GEF trust fund have resulted in disproportionality in burden sharing in all three previous replenishments.

Given this disproportionality, the question is what the implications thereof are for the issue of free riding and the undersupply of the public good. More specifically, can the findings on the observable burden sharing provide supportive evidence of the existence and extent of inefficiency? Furthermore, it is of interest to know who is alleged to free ride and how free riding can be reduced.

To analyze these questions, I first make some theoretical considerations in that I define a free-riding rule in an efficient outcome of collective actions. The derived benchmark can be used for a comparison with the actual burden sharing observed. However, a theoretical benchmark cannot be generalized. It depends upon the assumptions made regarding endowments, preferences, and technologies in the donor countries.

A Theoretical Benchmark: Burden Sharing in an Efficient Allocation

For the discussion, I reconsider the standard model introduced in Section 4.4.2.1. Donor countries display different levels of income but *identical preferences* of the Cobb–Douglas type.

If the within-ally burden is defined as $b_i = g_i^* / Y_i$ and $b_j = g_j^* / Y_j$, the question is whether in an efficient allocation $b_i > b_j$ or $b_i < b_j$ when $Y_i > Y_j$. In the former case, efficiency requires the high income country to carry a disproportionately large burden. In the latter case, this holds for the low income country and corresponds to the empirical findings.

Multiple combinations of the individual contributions, g_i^* and g_j^*, satisfy the *efficiency* condition in equation (4.22). Substituting g_i^* and g_j^* by using the definition of the within-ally burden, I derive a linear function that describes the set of burden-sharing allocations that simultaneously represent efficient allocation:

$$(4.48) \quad b_i = (1-\alpha)\left(\frac{Y_i + Y_j}{Y_i}\right) - \frac{Y_j}{Y_i} b_j .$$

Accordingly, examples of efficient allocations with $b_i > b_j$, as well as with $b_i < b_j$, can be found. This implies that the empirical result on the burden sharing depicted in Table 24 neither rules out free riding nor provides any supportive evidence of inefficiency.

Since multiple efficient allocations are attainable, a political decision is to be made as to which allocation, with its connected utility levels of the countries involved, is regarded as the most equitable. In Section 4.4.2, I introduced a simple welfare function that is linear in the utility of each of the two countries. The welfare aggregation is essentially described by the weighting parameter, λ, with $0 < \lambda < 1$, which indicates the importance assigned to the well-being of the high income country relative to the well-being of the low income country. Based upon this, I derived the contributions in the *social optimum* in equation (4.25).

Using these results, I now define the burden measures, b_i and b_j, in the social optimum as

$$(4.49)\quad b_i = 1 - \lambda\alpha\left(\frac{Y_i + Y_j}{Y_i}\right); \quad b_j = 1 - (1-\lambda)\alpha\left(\frac{Y_i + Y_j}{Y_j}\right).$$

When comparing the two terms for burden sharing and solving for λ, it turns out that, given that $Y_i > Y_j$, the social optimum only requires the high income country to carry a disproportionately large burden, $b_i > b_j$, if $\lambda < Y_i/(Y_i + Y_j)$. In other words, a socially optimal allocation with a shown disproportionality to the disfavor of the high income country requires that in a social optimum, the well-being of the high income country is given less weight in the welfare aggregation than would correspond to its share in the total income.

Vice versa, the disproportionality to the disfavor of the low income country observed for the finance of the GEF would only correspond to a social optimum if the well-being of the low income country is given even less weight than would correspond to its income share. The latter case can be ruled out, since it obviously contradicts the intuitive notion of equity. Accordingly, the empirical results of disproportionality imply that (1) a social optimum is presently not attained and (2) the high income countries should increase their contributions in order to attain disproportional burden sharing that is consistent with a socially optimal allocation.[266] On this basis, the empirical result on burden sharing serves as qualitative evidence of the free riding of the high income countries.

Differences in Preferences

This implication rests upon a number of assumptions. In the following, the assumption of identical preferences for donor countries is relaxed, and how this influences the interpretation of the empirical finding of burden sharing is studied.

In practice, there are certainly differences in the extent to which the *external benefits* generated from global environmental protection are felt by the various donor countries. For example, geographical latitude and altitude can determine the extent to which a country is vulnerable to the impact of ozone depletion or a sea level rise induced by climate change (Sandler and Murdoch 1997). Accordingly, the external benefit received from environmental protection, which is supposed to mitigate such adverse impacts, varies among countries. Furthermore, although large parts of global biodiversity are located in the developing world,

[266] An alternative hypothesis that states that the donor community makes excessive contributions to the GEF can be rejected by assuming that under common circumstances, donors that are sovereign in their decision making do not have an incentive to make contributions in excess of the benefits they receive.

some donor countries host substantial biodiversity endowments on their own territory as well, and the natural endowment of some donor countries is larger than that of others. If preserved domestic wildlife is considered a substitute for wildlife in the developing world and the utility a country derives from the existence of a marginal preserved species is positive yet decreasing, the biodiversity-abundant donor countries receive comparatively less additional benefits from investments in the conservation of global biodiversity.

If other things are held equal, the lower the benefit received from GEF-funded activities, the lower the incentive for the donor countries to provide financial resources. Nevertheless, several global environmental problems are lumped together in the GEF. Since each problem may be of different relevance to a single donor country and individual contributions cannot be earmarked for a particular global environmental problem, it is difficult to identify donor countries that have relatively weak or strong preferences for environmental protection, as supported by the GEF portfolio.

Considering the previous two-country two-commodity framework, suppose now that in addition to differences in the income level, there is a difference in preferences for environmental protection among donors. Suppose that the parameter α in the utility function changes to β for the low income country, where $\alpha \neq \beta$. It can be illustrated that the condition for efficient allocations (see equation (4.22)) transforms into

$$(4.50) \quad \beta g_i^* + \alpha g_j^* = (1-\alpha)\beta Y_i + \alpha(1-\beta)Y_j.$$[267]

Again, multiple *Pareto optima* for combinations of g_i^* and g_j^* exist. Using the definition of the within-ally burden, the geographic space of free riding allocation in Pareto-efficient outcomes is described by

$$(4.51) \quad b_i = (1-\alpha) + \alpha\left(\frac{1-\beta}{\beta}\right)\frac{Y_j}{Y_i} - \frac{\alpha}{\beta}\frac{Y_j}{Y_i}b_j .$$

The equation implies that Pareto efficiency can be satisfied for allocations that display disproportionate burden sharing in both of the two directions.

In order to abbreviate the analysis of the *social optimum*, I only provide an intuitive discussion: similar to the case of identical preferences, burden sharing in a social optimum depends upon the relative weight given to the well-being of

[267] Andreoni (1988) studies differences in the preferences in a two-commodity model framework of a private public good provision. Considering different types of utility functions, the author illustrates that in equilibrium only countries of one type—more precisely, the most generous type—make contributions. Countries of all other less generous types do not participate.

each country. In the same regard, the benefit each country receives from environmental protection influences the social optimum. It can be shown that in the case of identical preferences and equal weights in the countries' utilities, the high income country should carry a disproportionately large burden in financing environmental protection that reaches a socially optimal level. This disproportionality increases if the high income country shows a comparatively strong preference for environmental protection or decreases if the preference is comparatively weak.

Consequently, only when assuming that the high income country shows a weaker preference for preserving global environmental resources relative to consuming nonenvironment goods, i.e., $\alpha > \beta$, can it not be ruled out that the observed disproportionate free riding in the finance of the GEF corresponds to an allocation that is both efficient and equitable from a societal point of view. The extent to which this assumption holds true is partly addressed in the literature on the environmental Kuznets curve (see Section 2.3.4)—although studies in this context mainly focus upon investments in domestic environmental resources and not on expenditures for international transfers.

The Role of Unilateral Grants

Another assumption of the original model that needs to be reviewed is that environmental services of global importance are only financed by the GEF. In contrast, as shown for biodiversity and ecosystem services in Section 4.1.2.2, international donors also contribute on a unilateral basis. In this regard, a high income country may have a weak preference to contribute to the GEF because it prefers to make unilateral transfers for environmental protection rather than contribute to a multilateral institution that invests the money on behalf of a community of many donor countries (see Section 4.1.1.3).

For example, Kwon (1998) derives a disproportionality disfavoring the low income countries in the burden sharing of development finance (see also Addison et al. 2004). The author hypothesizes that this disproportionality results from the "special incentives" of the high income countries. More precisely, "as the United States and other big countries have less influence in [large multilateral organizations such as] the UN they become more interested in having organizations, like the G7, that will give more attention to their own concerns and that they control. Those high income countries that have come to contribute larger contributions of their GDP [to multilateral organizations like the UN]... —notably, the Scandinavian and the Benelux countries—are not big enough to be in the G7... . They may gain from international organizations in which numbers count and may be able to gain influence in such organizations by exceptional foreign aid" (p. 47).

Since the disproportionality identified in development assistance parallels the disproportionality observed in the finance of the GEF and since the GEF replenishment process to some extent relies upon the basic share framework in the IDA, which is a multilateral institution in development finance, the asserted hypothesis on the high income countries' behavior may also apply to the GEF.

Regarding the theoretical framework presented in Section 4.4.2, unilateral grants may be interpreted as perfect substitutes for contributions to the GEF in the financing of global environmental protection. Against this background, the free riding and equity issues have to be assessed on a different basis: if the type of the funding mechanism, i.e., multilateral versus unilateral, matters for a country's contributions, i.e., if a high income country prefers to make contributions outside the GEF, an indicated disproportionality disfavoring the low income countries in a GEF replenishment says little with respect to free riding and deviations from a socially acceptable allocation. Moreover, donors who carry a relatively small burden in the finance of the GEF may provide substantial contributions unilaterally. If this were the case, the observed disproportionality within the multilateral framework would ideally be offset by a (strongly) reverse disproportionality in the noncooperative framework.

I briefly examine this point by considering unilateral grants that OECD countries provide for environmental protection in the developing world. To describe the financial flows, I take data from the OECD DAC Creditor Reporting System (OECD 2004). I only consider flows of environmental aid for which a "Rio marker" applies, i.e., flows that refer to environmental issues addressed in the Rio conventions (see Section 4.1.2.2). The data refers to 1998, 1999, and 2000 only.[268] For 19 OECD countries, I calculate the sum of annual flows.[269] In order to describe the within-ally burden, the amount derived for each country is again divided by the country's GNI in current dollars (Atlas method). Following the procedures described in Section 4.4.3.2, I calculate the Spearman rank correlation coefficient for each year. Table 26 presents the results.

As indicated by the figures in parentheses, there is no statistically significant correlation. Moreover, the sign of the correlation is negative. This implies that the disproportionality derived in the finance of multilateral GEF is not offset by a reverse disproportionality in the unilateral finance of environmental protection. To resume, the analysis of the unilateral grants provides no counterargument for

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268 Although the considered three years altogether comprise 8,403 payment flows amounting to a nominal sum of $4.6 billion, the database is quite small in its temporal dimension. This is because for unilateral flows before 1998, Rio indicators have not (yet) been assigned and flows after 2000 are, to date, incompletely recorded in the OECD database.

269 The database does not list unilateral flows for Greece, Italy, and Luxembourg.

Table 26:
Rank Correlation between Within-Ally Burden and Income for Unilateral Donors

Year	Observations	Spearman rank correlation coefficient $\rho_{b\text{GNI}}$
1998	18	–0.174 (0.49)
1999	17	–0.265 (0.30)
2000	19	–0.095 (0.44)

Notes: b denotes the burden proxy. $\rho_{b\text{GNI}}$ denotes the Spearman ρ of the burden in unilateral actions to GNI. The figures in parentheses indicate the probability of a type I error when testing the null hypothesis of independence in a two-tailed test.

the finding that high income countries free ride (to some extent) in the financing of the GEF.

Additions and Conclusive Remarks

Aside from the modification discussed, the underlying model framework perceives environmental protection (including biodiversity conservation) as the only global public good. When broadening the playing field further, interaction in the provision of *multiple global public goods* can be considered (Boyer 1990). In this regard, disproportionality in the financing of global environmental protection may be offset by reverse disproportionality in the financing of other global public goods, such as preventing communicable diseases or reducing poverty. Moreover, some global public goods rely upon in-kind contributions, with the consequence that certain (high income) donors have a comparative advantage in their provision. For reasons of space, I do not discuss these aspects any further. Nevertheless, it should be borne in mind that the results derived tend to imply that particularly the high income donors should increase their contributions to the GEF in order to attain efficient and socially acceptable financing. This implication, however, neglects the fact that environmental protection competes with other global public goods for international funding.

Regarding the calls for the financial strengthening of the GEF, I ask how donors can be made to contribute more. Olson and Zeckhauser (1966), who have made comparable findings of disproportionality in the context of other global public goods, believe that "moral suasion is inappropriate, since different contributions are not due to different moral attitudes, and ineffective, since the less than proportionate contributions of the [high income] countries are ... grounded in the national interests." According to the authors, a way out that could lead to an increase in contributions is "to design policy changes that would leave everyone better off."

While such policy changes with respect to the finance of the GEF have seldom featured in the public discussion, some general recommendations to overcome collective action problems as summarized by Sandler (1992: 58ff.) may apply. The author, inter alia, proposes limiting the incentive to free ride by creating private joint products and selective incentives, i.e., the tying of private benefits for donors to environmental protection as the global public good concerned (Sandler and Hartley 2001). In Section 4.4.2, I briefly discussed potential joint products with private benefits that exist in the context of the GEF. Although free riding is reduced if joint products are allowed to play a more important role, it is unclear whether this leads to an increase in the supply of the public good, i.e., the total size of the GEF trust fund. Moreover, such institutional changes may conflict with transparency as one guiding principle in the replenishment process.

4.5 Summary of Results

In this chapter, I investigated how ecosystem services of global importance that are not traded on markets can be safeguarded by measures of international biodiversity policy. I focused upon ecosystem services that represent pure *global public goods* whose provision is supported by *protected areas* as a specific form of natural resource management. Ecosystem services that represent impure public goods, e.g., ecotourism services, were only considered insofar as they are generated as joint products.

Since ecosystem services with the character of a global public good are often generated in countries that do not have a self-interest in ecosystem protection, the international community arranges for mechanisms that create incentives for these countries to protect their natural resources to a globally efficient extent. Given the asymmetric distribution of global biodiversity, these mechanisms in particular address natural resources in developing countries. Moreover, since the countries have sovereign rights to their resources, the major element of these mechanisms is *transfers* that are granted conditional on additional efforts for the preservation of sites that host globally important biodiversity.

The analysis in this chapter proceeded in two parts. My starting point was a conceptual analysis of domestic and international protected area policies and an empirical description of the current outcome of these policies. Here, it turned out that the *Global Environment Facility* (GEF) plays an important role in this context. Accordingly, I studied specific questions on international transfers for protected areas against the empirical background of the GEF.

Conceptual Challenges to International Policies on Biodiversity and Protected Areas

I initially approached protected area policy on a conceptual basis. I described that a domestic policy can include different combinations of private, communal, and state property rights with respect to land and the natural resources hosted therein. The property rights regime determines the set of instruments that is available to implement a natural resource management that is efficient from a domestic perspective. Domestic policymakers typically disregard international spillovers of this policy.

I argued that summation or weighted sum technologies describe the collective provision of biodiversity as a global public good in the most appropriate way. Findings in the economic literature imply that for both *aggregation technologies*, the provision of the public good falls short of the optimum from a global perspective. I applied these results to the collective actions on biodiversity conservation, which in effect suggested that public-good-like ecosystem services of global importance are undersupplied. A Pareto-improving international policy in this regard has to account for the directions of biodiversity spillovers. Studies in the natural sciences argue that although there is virtually a complex network of ecological interdependences around the globe, unilateral spillovers from resource countries in the developing world to the developed countries dominate the global flows of biodiversity externalities. For the economic internalization of these flows, developed countries ideally implement reverse flows of *transfer* payments.

Several economic studies deal with negotiations on transfers and international environmental protection. They identify several incentive problems that can occur in this context and inhibit an optimal negotiation outcome. Considering the negotiations on transfers and measures to protect global biodiversity, I applied the findings of these studies to describe the interactions between resource countries as the owners of biodiversity and donors in the developed world as beneficiaries thereof. I suggest that *incentive problems* occur because of private information regarding the benefits and costs of biodiversity conservation and because of the irreversibility of ecosystem change, which, under certain circumstances, offers resource countries the chance to behave strategically. I briefly recapitulated the means to mitigate the impact of the incentive problems as discussed in the literature. These means relate to an appropriate design of negotiation process and transfer payments. In spite of mitigation, incentive problems are likely to inhibit the attainment of an agreement on the globally efficient level of conservation.

Within the international donor community, countries coordinate within a *multilateral framework* or, alternatively, act on a *unilateral basis*. I discussed that the choice of each donor country depends upon the specific characteristics, such as the exclusiveness of the transboundary spillovers generated from a resource country's policy or the special relationship and ties of a donor country to a resource country that can be helpful in negotiating a contract for effective protection.

Assessment of the Previous Protected Area Policy

Regarding the transfers in a coordinated multilateral framework, the empirical evidence presented implies that the *CBD*, with the *GEF* as its associated financing mechanism, is of particular importance. *Unilateral grants* are primarily provided in official arrangements. In order to assess the transfer flow in quantitative terms, I analyzed data taken from official databases and found that, in a conservative estimate, total unilateral grants amount to an annual size similar to the total annual grants by the GEF. In an alternative classification, unilateral grants are up to four times larger than those supplied by the GEF and other multilateral financing institutions. In addition to official spending, nongovernmental organizations also provide international funding on a unilateral basis. Based upon results of an official study, I found that the current amount of *private giving* is notably smaller than that of official spending.

It turns out that the transfers made for the purpose of biodiversity conservation in many cases do not directly address protected area management, but assist other instruments in biodiversity policy. Since the flows often cannot be differentiated in this respect, the figures I derived on transfer flow size represent the upper limit of the actual funding for protected areas.

I described that the international biodiversity policy has formulated both quantitative and qualitative objectives concerning *a global network of protected areas*. Although the total number of protected areas has been expanded in recent decades, official studies still identify gaps in the current global network in order to consider a globally representative biodiversity to a sufficient extent. Furthermore, domestic efforts of the resource countries are often insufficient to guarantee the effective management of the designated protected areas.

I reviewed estimates on the *financial demand* for safeguarding the global network of protected areas. These estimates derived substantially exceed the financial resources currently invested in the management of protected areas worldwide. I argued that the responsibilities of international donors in this regard cannot be described in a precise way, since the protected areas also generate domestic benefits and are also (partly) financed by domestic means. Yet, my analysis confirmed a shortfall of financial resources, which suggests that the current size of international transfers is below the level necessary to secure globally optimal protection.

The remainder of this chapter conducted an empirical analysis of specific questions on international transfer policy for protected areas. Because of its prominent role in international biodiversity policy and its comparatively well-described funding activities, I chose the GEF as the object of analysis.

The Way the GEF Arranges for Biodiversity and Protected Area Management

I investigated where the international funding goes and, given that the GEF aims at financing biodiversity conservation, what this tells us about the actual costs and benefits of protected area management. The GEF biodiversity portfolio served as the empirical basis of my analysis: more than 650 projects that have been approved in this GEF focal area from 1991 to 2004 were included in my study.

I demonstrated that small projects, such as *enabling activities*, which primarily support future domestic policies in the resource countries, are largely financed by the GEF. In contrast, *medium-size and large projects* display a comparatively lower funding share. Domestic stakeholders and other international donors participate as cofinancing parties.

Regarding the allocation of GEF projects and project funding across the various regions and *continents*, it turned out that most of projects and most of the aggregate funding goes to Africa. A smaller but still very large share of the GEF funding is invested in Asia, Latin America, and the Caribbean. Fewer transfers are directed to transition countries in Europe and Continental Asia and transboundary projects.

Considering the recently formed interest group of the *LMMCs*, i.e., countries with an exceptional natural abundance of biodiversity, I discovered that the allocation of GEF funds is not biased towards these countries. Moreover, funding by the GEF is broadly distributed across resource countries with an abundance of biodiversity of various sizes.

In the same way, the GEF does not seem to favor the *very poor countries* among the signatories of the CBD. I demonstrated that the allocation of projects across countries of different classes of national income more or less corresponds to the distribution of the CBD signatories across the same classification. In contrast, I found suggestive evidence that GEF biodiversity projects are carried out primarily in resource countries that have a good or satisfactory *governance structure*. Countries with bad governance attract comparatively low project funding.

GEF biodiversity projects are assigned to operational programs that are defined by the needs of specific ecosystems and, hence, are related to ecological criteria. The analysis showed that within the sample of projects, *forest ecosystems* attract most of the total GEF funding, followed by *coastal, marine, and fresh water ecosystems*.

The biodiversity projects usually address multiple objectives that all relate to the goal of conservation or the sustainable use of biodiversity, but they also take other social objectives into account. In particular, project actions try to mitigate potential conflicts between environmental protection and other objectives, especially economic development, poverty alleviation, and/or the acknowledgement of the traditional rights of indigenous people. It turned out that in order to achieve their various objectives, the projects make use of *a bundle of instruments*. Protecting natural areas is only one of them—although an important one.

When regarding the projects on terrestrial ecosystems and concentrating upon those projects that include protected area measures, the study revealed that the establishment of *new sites* or the *expansion of existing ones* is only included in the course of action for a minority of the projects. Moreover, the internationally funded activities strongly focus upon supporting an effective management regime in already designated sites.

Activities in protected areas that obtain GEF support also attract substantial funding from other international sources. This funding is mostly provided on a grant basis, although some funding is provided on a loan basis. I calculated and described average shares in the project funding for various classes of international and domestic donors.

Effective protected area management demands continuous management input, which generates a continuous demand for financial resources. Since the GEF projects only last for two to ten years, institutions that ideally guarantee the *long-term* and *sustainable financing* of protected areas have to be implemented. In this regard, I concentrated upon the role of ecotourism and environmental funds; I briefly discussed these two mechanisms and analyzed how they are supported in the GEF projects.

I demonstrated that *ecotourism* is frequently part of project actions aimed at the integration of biodiversity conservation and local community-based development. Empirical evidence from selected projects indicated that GEF assistance typically has the character of an investment that generates future proceeds in the form of benefits from conservation or revenues that can be (re-)invested in conservation. Considering *environmental funds*, I illustrated that they are only addressed in certain projects. The GEF either participates in the capitalization of national environmental funds or assists in the implementation of environmental funds that are still used increasingly in protected area management.

I concluded that the flow of funding generated by both mechanisms is linked to fluctuations in aggregate demand and/or international interest rates and is, therefore, subject to uncertainty regarding future economic development. In addition to site-specific circumstances, it thus remains unclear as to whether the mechansims can actually secure long-term financial sustainability in a specific case.

The Way the GEF Donors Attempt to Secure the Cost-Effective Use of Their Transfers

The question of cost-effectiveness is related to the application of the *incremental cost principle*. This principle is applied to calculate the grant amount the GEF ideally provides the resource countries to maintain biodiversity of global importance.

The analysis of the incremental cost principle in this chapter encompassed three areas. First, I recapitulated the *legal formulation* of the principle, its refinement in the full, agreed upon incremental cost approach, and its implementation in a project-specific context. I reproduced recent appraisals stating that the principle is generally "left in vague terms" (Cervigni 1998) in order to reconcile the *diverging interests* of the resource country, as the project host, and the GEF donor community: while the donor community aims to attain the maximal quality of conservation for a given amount of financial resources (or to attain a given conservation target at minimum costs), the resource country has an interest in using its natural resources in order to maximize its own well-being either by depleting its resource stock or by exhausting the international willingness to pay for resource preservation. In negotiations, the two parties have to find a way to alleviate the conflicts that result from these diverging interests. This constitutes the general background of the policy of the incremental cost principle.

In addition to the proper *definition of the baseline* upon which the GEF-supported conservation activities are built, negotiations deal with *domestic benefits* that are generated from these incremental activities. On the one hand, the deduction of any of these benefits from the transfer amount provided by the GEF is regarded as a requirement for the cost-effective operation of the GEF mechanism; on the other hand, developing countries as recipients may have few *incentives to participate* in negotiations if they have little to gain from incremental conservation.

Against this background, I continued with a survey of the theoretical literature on international transfers for conservation. I compared economic models that describe international transfer policy on a theoretical basis. Although the incremental cost principle was not subject to investigation in all of the models, they were suitable to study the impact upon (1) the transfer size of taking *domestic benefits* of incremental conservation into account to different degrees and (2) of the *role of bargaining power* for both the agreed upon level of conservation and the size of the transfer.

Results from a simple *partial equilibrium* model indicate that while the incremental cost principle as an institutional rule cannot guarantee an allocation that is per se efficient, an efficient outcome may fulfill the requirements of the principle. Furthermore, when applying the theoretical results to the practical provisions of the GEF, it is implied that, under certain conditions, the incremental cost principle reduces the set of possible outcomes. In particular, the principle excludes agreements that arrange for transfers in excess of the incremental expenditures but that nevertheless yield a Pareto improvement for the donors.

I described how the literature uses *dynamic models of natural renewable resources* in order to analyze different types of transfer regimes with different bases for transfer payments. When applying the theoretical findings to the actual GEF transfer scheme, it is implied that the incremental cost principle does not address stock-based transfer payments but only considers payments for conservation at the margin, i.e., the set of feasible solutions to the negotiation problem is once more limited by the principle's provisions.

Finally, I recapitulated a two-country two-commodity model by Cervigni (1998). The model describes how *efficiency* and the *equity* issue of biodiversity conservation are interrelated: on the one hand, any agreement that takes the incremental domestic benefits into account determines the size of the surplus that a resource country receives as the project host; on the other hand, the extent to which the incremental domestic benefits are taken into account influences the effectiveness of the transfer payments, i.e., the relative prices of conservation from the point of view of the donor community. Accordingly, whenever the incremental domestic benefits are only taken into account to a small extent, the welfare of the transfer recipients tends to increase, while the willingness of the transfer donors to pay, and, thus, the amount of financial resources they make available for conservation purposes, is reduced.

In the final part, I studied the *empirical implementation* of the principle in incremental cost analyses and assessments. By using a sample of selected GEF biodiversity projects, I analyzed how incremental domestic benefits are treated in *incremental cost assessments* contained in official project documents.

The results showed that in the regular case, incremental domestic benefits are *not quantified*, but only described in qualitative terms. In the exceptional cases where the benefits are estimated in monetary units, they amount to the size of one or two thirds of the total expenditures for incremental conservation.

Regarding the *average financing* of incremental activities as represented in the documents, it turned out that the GEF carries the costs for approximately

68 percent of the activities. International multilateral and unilateral assistence finances another 4 percent and 6 percent. The participation of domestic financiers in covering the remaining costs indicates that these activities indeed generate incremental domestic benefits that are at least partly accounted for when determining the size of the international transfer.

Based upon the incremental cost assessments in the *individual project documents*, I studied figures on the international and domestic funding of incremental activities. I discussed whether these figures can explain the size of incremental domestic benefits and, thereby, the extent to which they are taken into account when determining the transfer amount. Given that an incremental domestic benefit may not be generated in every single case and that the benefit is generally difficult to identify, the projects in the study sample were sorted according to the two concepts of the net incremental cost and gross incremental cost approach.

Although approximately one third of the projects displays a complete financing of incremental conservation by the GEF, the empirical findings indicate that in the majority of cases, incremental activities show mixed domestic and international financing. Based upon some reasonable assumptions regarding the existence of incremental domestic benefits and the extent to which they are taken into account, this result suggests that, in most cases, the parties compromise between the net and gross incremental cost approach, which also implies a compromise with respect to strict cost effectiveness.

Finally, I analyzed whether the GEF share in the financing of incremental activities changes when considering projects in different groups of resource countries. The recipient countries were classified according to criteria that hypothetically explain the resource country's *bargaining power* in the negotiation process with the donor. Overall, the empirical findings show that the share of the GEF does not change significantly across the country groups. Consequently, my analysis did not find evidence of a differentiated bargaining power within the group of resource countries.

The Finance of the GEF Trust Fund: Evidence on Free Riding and Burden Sharing among Donors

The considered international transfers to resource countries in the developing world can be perceived as side payments to make them join cooperative actions for the preservation of globally important environmental resources, including biodiversity. The GEF as an institutional regime that provides transfers in the multilateral framework itself represents an *intermediate public good* that assists global environmental protection.

Against the background of recent *calls for the financial strengthening* of the GEF, I described how international donors arrange for the financing of the GEF and studied whether I could find evidence of free riding among donors and how equity in burden sharing is arranged. The underlying question refers to the perspectives of a future increase in the size of the GEF trust fund.

I initially described the *GEF financing arrangement*, showing that, in addition to major developed countries, transition and developing countries that are eligible to receive GEF grants also participate in the financing. The proportions among the contributions of developed, or, synonymously, nonrecipient, countries are governed by the burden-sharing scheme that is applied in the multilateral financing of the International Development Association (IDA). The funding by individual donor countries cannot be earmarked for specific focal areas such as biodiversity. Considering the size of the individual contributions, I found that certain donor countries merely fulfill the given income-inelastic minimum requirement, while other countries make contributions in excess of this minimum amount or the contributions derived from the burden-sharing scheme.

Based upon data in official documents, I demonstrated that in the recent replenishments, the *total size* of the GEF trust fund (in terms of current dollars) has been increasing steadily. The seven major industrialized countries, i.e., the G-7 group, provided approximately three quarters of the fund's endowment. Several newly industrialized countries and transition countries have not yet joined the replenishment arrangement or have withdrawn.

In order to analyze the interdependencies in the financing of the GEF, I continued with a survey of theoretic models of private public good provision and related the findings of the various models to the current GEF arrangement. Because of the small data sample, I only conducted a qualitative analysis and derived some suggestive evidence.

First, I related the findings of the *one-period collective action model* to the question of the *behavior* that can be assumed for GEF donors. In other words, I analyzed the extent to which the donors' behavior expressed in the replenishment process matches the types of behavioral assumptions that are classified in economic theory. I found some evidence that suggests that the donor countries coordinate as implied by Lindahl behavior and, thus, that the size of the basic contributions within the subgroup of nonrecipient countries is less subject to strategic interactions. In contrast, burden sharing between the groups of nonrecipient countries and recipient countries displays certain features of a leader-follower relationship that suggest that nonrecipient countries have a strategic advantage. Nevertheless, it became clear that in order to support these two hypotheses, further research is needed.

When using a theoretical one-period setting to explain the arrangement for the GEF replenishment, I abstracted from *dynamic incentive problems* that have an impact upon the current observable agreement. More precisely, I recapitulated theoretical studies on repeated interactions that suggest that (over a certain range) an increasing contribution by one donor country may stimulate incentives for the remaining countries to increase their contributions as well. In other cases, it reinforces the incentives for other donors to deviate from their current commitments or provide smaller contributions in an upcoming replenishment. In the same context, it increases the incentives for outsiders not to undertake a commitment to future contributions. I related the general findings to the financing of the GEF: a donor country may anticipate the adverse reactions of the other donors if it unilaterally increases its GEF contribution. In this respect, any agreement on GEF contributions is not only based upon the donor countries' actual preferences for environmental protection but is also influenced by the aim to sustain the existing cooperation within the donor community and, thereby, at least guarantee funding to the total amount that is currently provided.

Furthermore, the literature shows that other parameters, such as the relative size of national income, can have an impact upon the strategic behavior of the potential donor countries. With respect to the GEF, the results suggest that *income inequality* reinforces the incentives to free ride and, hence, inhibits reaching a self-enforcing agreement with an enlarged donor community or with increased total funding in the future. Finally, a comparison of the literature and the empirical description of the GEF suggested that side payments to overcome incentive constraints within the donor community are apparently not provided for in the current institutional framework.

Regarding free-riding behavior, countries that do not make contributions to the GEF, although it can be expected that they receive external benefits from GEF-funded activities, can be distinguished from countries that participate in the financing of the GEF but spend less in comparison to the benefits they receive. I did not empirically study the first type of free riding. Nevertheless, I did provide evidence that would seem to suggest that nonparticipating countries that have a higher income than small DAC countries (such as Luxembourg, Ireland, and Portugal) tend to free ride. I pointed out, however, that such an implication would disregard potential differences in preferences for environmental protection including biodiversity conservation.

Considering the second type of free riding, I studied the *burden sharing* in previous GEF replenishment processes and discussed whether the results derived can provide evidence with respect to inefficiency and free riding. With respect to fairness, I demonstrated that the burden sharing in the previous replenishments satisfied the definition of weak vertical equity. Regarding differentials in contributions relative to differentials in the national income, I found out that there is a disproportionality disfavoring the low income donor countries. This disproportionality parallels the observed disproportionality in the financing of development assistance.

I used the empirical findings on equity in my own 2x2-model framework to show that multiple efficient allocations exist, each connected to a different burden-sharing allocation. Accordingly, I could not identify free-riding behavior in this regard. Nevertheless, given reasonable assumptions regarding a social optimum, the findings indicate that this optimum is not currently attained and that the contributions of the high income donors fall short of the optimal level.

Finally, my theoretical discussion assumed that the size of the GEF trust fund approximates the provision of environmental protection as a global public good. However, *unilateral grants* provided for activities that generate globally important environmental services might serve as substitutes for GEF grants. In order to put the burden sharing in the financing of the GEF into a broader context of international assistance, I also investigated the burden sharing in unilateral actions. The empirical results again indicated disproportionality to the disfavor of the low income donors. Consequently, the disproportionate burden sharing in the multilateral framework is not compensated in the noncooperative unilateral framework.

5 Conclusion

In order to conserve globally important biodiversity, sovereign countries have to agree upon an international regime of natural resource management that is both effective and efficient. Several mechanisms are available for this purpose. The study at hand concentrated upon two *selected policy instruments*: first, the commercial use of, and trade in, genetic resources, which represents a property rights approach to biodiversity, and, second, the use of international transfers for protected area management, which represents a combination of the international price-based regulation of biodiversity use and domestic command and control measures.

The core questions regarding these instruments refer to *criteria* that are traditionally used to evaluate environmental policy instruments:

- How can we assess the environmental effectiveness of each of the two instruments? Does it effectively contribute to the maintenance of globally important biodiversity?
- How can we assess economic efficiency? Does the instrument assist the achievement of conservation in a cost-minimizing fashion? Does it support an allocation that leads to an overall Pareto improvement? And does it stimulate incentives to develop improved resource management in the long run?
- How can we assess the distributional effects resulting from the instrument? Does it distribute costs and benefits in such a way that policy intervention is widely accepted and, thus, effectively enforced?

These questions, together with the question concerning administrative feasibility, have been the subject of several studies in economics and other sciences. Their findings imply that, although the two instruments have their strengths, neither of them can provide conservation that is sufficient from an international perspective. Yet, each of them can be part of a policy mix that is applied in a region- or site-specific context. Furthermore, the studies discuss whether the instruments can be developed in order to improve their effectiveness. With regard to these two aspects, the conceptual and empirical findings of my study provide additional and new information that is useful for the process of policymaking.

Methodologically, I used and generated empirical data for descriptive analyses and applied or developed simple analytical models that are suitable for dealing with the identified research questions. The conceptual background of the study is determined by the fact that biodiversity-abundant ecosystems jointly provide bundles of valuable economic services. These services either represent public goods or are defined as a private good. For the design of the policy instruments, the key feature is that resource management can *jointly* generate private and public goods, as well as local and global public goods.

The study focused on the preservation of ecosystem services with a *global public good* character. Genetic resources as valuable private goods are extracted from natural habitats that jointly preserve biodiversity as a public good of global importance. The commercial use of and trade in genetic resources in this regard create incentives for private habitat preservation. Furthermore, biodiversity-abundant developing countries use protected area policy to maintain ecosystem services that represent local public goods. This policy also generates global public goods that are, however, not addressed by domestic policymakers. Since the global public goods are, therefore, undersupplied, international transfers aim to create incentives for developing countries to internalize the spillovers of their policy. Finally, both parts of the study addressed *land use change*, which is the most important direct driver of biodiversity loss and ecosystem change.

Market-Based Incentives to Preserve Biodiversity: Trade and Commercial Use of Genetic Resources

Considering the creation of markets for genetic resources to be an instrument of conservation, policymakers often presuppose that *environmental effectiveness* is given. This held true particularly for the negotiation and signing of the Convention of Biological Diversity (CBD). Recent findings in economics and other social sciences, however, have challenged this point of view. Against this background, I focused my analysis upon the issue of environmental effectiveness.

Since markets and trade interactions demand well-defined *property rights*, I first recapitulated the current debate on property rights to genetic resources and its embodied information. I identified and described four key issues that have an influence upon the allocation of natural resources and resources in R&D. This is, first, the relationship between intellectual property rights (IPRs) and wild genetic resources and, second, the use of IPRs to protect indigenous knowledge regarding the use of biodiversity and/or genetic resources. Third, it is the impact of IPRs for biotechnological goods and, finally, the nexus between the access to technology and biodiversity and IPRs. I briefly described each issue upon a conceptual basis and summarized the international regulations that are currently relevant. Since these findings relate primarily to wild genetic material and its embodied information, I also summarized the major particularities of property rights to plant genetic resources for food and agriculture (PGRFA). Overall, I demonstrated that markets are only established for some but not all types of genetic resources. Recent policy efforts both in developed and developing countries aim to reinforce the excludability of the economic use of both genetic material and embodied/related information.

Considering the relationship between the establishment of markets for (in situ) genetic resources and the creation of (private) incentives for habitat conservation, *relative prices* for genetic resources influence the decision on preservation as one of alternative land uses. I illustrated this connection through the use of two simple analytical frameworks. Some frictions can prevail under certain circumstances and impede the specific environmental effectiveness of the market mechanism. I identified these frictions from a review of the literature: potential impediments include information problems and extensive transaction costs of market transactions with genetic resources. Furthermore, in certain cases, bioprospecting habitats may only generate public ecosystem services to a negligible extent.

The analysis of environmental effectiveness concentrated upon the traded quantities and associated market prices. Information on these two parameters is essential to assess the contribution of the market to biodiversity conservation, since it provides evidence of the revenues that the owners of biodiversity can obtain. Only if the owners receive sufficient revenues from conservation will they have an incentive to make investments for the effective conservation of natural habitats.

In an initial step, I provided an descriptive analysis of commercial use in the different industrial sectors based recent studies, but with updated and extended information. I indicated that genetic resources display different characteristics across the industries, which corresponds to the different quantitative and qualitative material demand in the individual sectors. For this reason, I concluded that the impact upon conservation resulting the trade in genetic resources has to be distinguished according to sectoral uses.

It turned out that for the industries with the largest market size, commercial uses of genetic resources are characterized by *uncertainty* and *nonrivalry*. Given the fact that specific genetic information is not bound to a specific material and that identical genetic material is frequently widely distributed, the value of specific genetic information is not linked to a specific in situ habitat. This complicates the creation of effective incentives for conservation and, thereby, reduces the contribution of markets to environmental effectiveness. The combination of nonrivalry, uncertainty, and *supply side competition* suggests that the market price for genetic resources is comparatively low. My analysis provided some evidence that supports this assessment. Accordingly, land use other than preserving in situ habitats is likely to be more profitable from the point of view of the landowners. For policymakers, this would imply that the effectiveness of the market-based strategy for conservation is rather modest and commonly cannot play a leading role in biodiversity policy.

Both the supply and demand for genetic resources, as well as for commodities from alternative land use, may change over time. This would have an influence on the impact of the market upon conservation. Some general assumptions can be made about future development:

Although I did not investigate trends in the *opportunity costs of conservation*, empirical evidence of tropical deforestation, for example, suggests that in many places these costs are strongly increasing. Against this background, reinforced property rights to in situ genetic material to enable resource countries to capture greater benefits from their natural endowments create few incentives for conservation. In a favorable case, a reinforced legal position of the resource owners may lead to an increase in the price of genetic resources in absolute terms, yet its relative price is in many cases still too low to render conservation profitable. Ceteris paribus, biodiversity degradation continues independently of corrections of potential property rights and market failures for genetic diversity as an economic good.

However, *prices for genetic resources* may also increase in the future. If an increase in market prices sustainably exceeds the assumed increase in conservation costs, conservation may one day become the preferred land use. The focus of my study, as well as of the literature, falls upon industrial demand for genetic resources in the pharmaceutical sector. In this context, no evidence of the increasing scarcity of genetic material and increasing market prices was found. However, other industrial sectors also act in the market and demand genetic material. While this demand is presently considered modest in quantitative terms, it may rise if the technology and market in these sectors develops. The impact of the market upon conservation may change accordingly. Based upon my findings, it can be assumed that even though certain trends can be identified in the market for genetic resources, this market is still in flux and, consequently, potential future development needs to be taken into account when assessing its contribution to conservation.

The differentiated properties of the commercial use of genetic resources that I described also imply that an assessment of environmental effectiveness cannot be generalized on the basis of aggregate data, anecdotal evidence, and stylized properties of the market. Moreover, a precise assessment of the described market-based strategy for conservation demands investigation on a case study level.

Efficiency is a further criterion to assess the market-based strategy to conservation. The literature suggests that trade in genetic resources leads to a Pareto improvement for the stakeholders. However, the trade in these private goods cannot provide for a *Pareto-efficient allocation* of biodiversity as long as the market mechanism does not enable the owners of ecosystems to capture the values of biodiversity beyond the immediate resource values. This relates to the nonuse values of biodiversity, including the values for future generations. Furthermore, efficiency is generally related to the proper definition of property rights for an economic good and its sound use. I briefly addressed these aspects of efficiency in the section on property rights to genetic resources.

Finally, the efficiency of a market-based strategy for conservation relates to the creation of *dynamic incentives* for improvements in resource management that lead to improved environmental quality in the long run. Considering the incentives for the long-term conservation of natural habitats that serve as bioprospecting sites, my analysis yielded results that suggest that markets for genetic resources are hardly capable of creating dynamic incentives in this regard: before developing any improvement in the resource management of a bioprospecting site, the owner of the site must be able to receive a sustainable flow of income from a market supply of genetic resources. This requires the commercial users of these resources to repeatedly conduct on-site collections at a specific site. Based upon the empirical description in the study, this condition applies to the conventional use of genetic material but not its informational use in the major industrial sectors. In the latter case, the relevant information can be reproduced in ex situ facilities, i.e., the user does not need to purchase any further in situ genetic material.

As regards the continuing demand for genetic resources from one specific site, the question is whether the owner of the site has an incentive to invest the market revenues received from in situ conservation. Whenever demand ceases after the first collection and the owner cannot contract future bioprospecting activities on his site, it is profitable for him to use the revenues resulting from the first contract for purposes other than conservation and convert the site.

Nevertheless, the stock of in situ genetic information is changing due to evolutionary forces. In this respect, the long-term preservation of natural habitats can serve to create new, naturally occurring genetic information. This could be of interest to commercial users. It is conceivable that the creation of in situ genetic information as a specific ecosystem service can be brought about in a market framework. In this regard, commercial users may have an incentive to pay for the long-term preservation of a protected bioprospecting site. However, I identified evidence that, in practice, the public sector jointly provides this ecosystem service through the management of state-owned protected areas.

The distributional effects of trade in genetic resources are closely related to the *equitable sharing* of the surplus of genetic information between the supply and demand side of genetic resources. This sharing of the surplus influences the political feasibility of international trade in genetic resources. In particular, the resource countries' acceptance of an outflow of (in situ) genetic information depends upon how benefit sharing is arranged. The *access and benefit sharing* *regulations* in the resource country represent the major instrument in this respect. I summarized the discussion on this issue and described that compared to the pre-CBD situation, the demand side has to fulfill additional requirements for information and transparency that in effect strengthen the position of the supply side in bilateral bargaining over the exchange of genetic materials. In this context, the literature often assumes that reinforced national laws on genetic resources influence the market value of genetic resources and, therefore, have an impact upon the incentive to preserve natural habitats. Nevertheless, as shown by my analysis, this impact is sensitive to the overall scarcity of specific genetic resources. Even with effective access and benefit-sharing provisions, the owners of biodiversity do not receive more revenues if the demand for genetic material is low and the supply side is characterized by intensive competition. Yet, the literature argues that with these legal provisions, partnership in trade is generally becoming fairer. This apparently holds true in spite of continuing allegations in the public of *biopiracy*, i.e., commercial users are accused of acquiring genetic material, information and associated indigenous knowledge in a way that is not consistent with the CBD provisions on access and benefit sharing.

I only briefly discussed the *administrative costs* of market transactions in genetic resources. Transaction costs incurred by the trading partners in the market are distinguished from transaction costs incurred by the public sector, which promotes the establishment of the market, for example, by reinforcing property rights and monitoring compliance with access and benefit-sharing provisions. The latter type of transaction costs relates to the question of the efficient use of public funds for alternative biodiversity instruments. If administrative costs for the public sector are substantial but the efficiency and environmental effectiveness of markets for genetic resources is rather modest, other policy instruments are likely to be more favorable. Any assessment in this regard has to be made on a site-specific level.

In addition to the policy-oriented implications, my study provides some insights that could be useful for the academic investigation of the market for genetic resources: estimates on the market value of genetic resources are often based upon analytical descriptions of R&D processes, including specific assumptions regarding the ecological and economic characteristics of the genetic material used. I described how the individual analytical studies emphasize the specific properties of the commercial uses of genetic material while neglecting others. I discussed how the diverse characteristics assigned to genetic resources in models lead to varying results on the private resource value. Future research could concern itself with developing an integrated modeling framework to define scenarios consisting of various combinations of characteristics and to study the varying impact of a market for genetic resources on conservation.

Preserving Biodiversity as a Global Public Good: Protected Areas and International Transfers

In the second part of the study, I dealt with international transfers for protected area policies in biodiversity-abundant developing countries. In contrast to the first part, in which I primarily studied private ventures in the conservation and sustainable use of biodiversity, the second part focused on intergovernmental bargaining and coordination between sovereign countries.

Considering the *environmental effectiveness* of protected area policy from an international perspective, the academic discussion typically asks whether protected area policy can effectively contribute to attaining the current political objective of a deceleration in the loss of global biodiversity by 2010. In this regard, I concentrated on the nexus between environmental effectiveness of protected areas and international transfers: since protected areas generate several non-market ecosystem services, funding from external sources is needed in order to maintain and improve the management of the sites already designated, as well as to expand the global network of protected areas. Since large endowments of global biodiversity are located in developing countries, developed countries need to mobilize a substantial part of this external funding as international transfers.

I recapitulated the conceptual background of international transfers for biodiversity conservation and presented empirical figures on the current aggregate transfers. The findings suggest that international assistance, described in current dollars, has increased slightly in recent decades but still remains at a modest level compared to the amount estimated to be necessary to manage an optimal global network of protected areas. In more detail, I empirically analyzed the use of the funding that the Global Environment Facility (GEF) provides for biodiversity conservation. I discovered that GEF-funded activities in most cases support the management of sites already designated. In this respect, the transfer policy in its current definition does not necessarily lead to a spatial expansion of the network of protected areas. These two findings may generally support some skepticism regarding the environmental effectiveness of the current arrangement of protected area policy and international transfers. However, the discussion on the estimated costs of an optimal global network suggests that further research is needed to quantify the gaps in the current network that have to be filled in order to support the 2010 target.

The *efficiency* of protected area policy can be analyzed on different spatial levels. Considering the local level of protected area policy, I briefly summarized the ecological-economic literature on the issue of efficient *reserve design*. Furthermore, I illustrated that the protected area policy on a national level can be designed in alternative ways with different combinations of private and public property rights to natural areas and the biological resources hosted therein. Although the various designs of protected area regulation can be the subject of a normative analysis, this was outside the scope of the study.

I addressed a finding in the literature stating that *domestic policy distortions* in resource countries is a serious problem for efficient biodiversity conservation, since they create perverse incentives for ecosystem use on a local level. These distortions also influence coordination on an international level. I presented suggestive evidence from the empirical analysis of biodiversity projects cofinanced by the Global Environment Facility (GEF) that indicates that developing countries with bad governance receive comparatively fewer international transfers. In this respect, I have contributed to the growing literature on governance and biodiversity policy in resource countries by highlighting the relationship between domestic governance and international environmental assistance.

International transfers involve negotiations on biodiversity conservation in excess of the level that is desirable from the resource country's perspective. The results of the conceptual studies imply that, since both the resource country and the donor country are sovereign in their decision making, any contractual agreement on extra conservation and transfer payment is *Pareto improving*.

Considering the interactions in the negotiation and enforcement of a concluded contract, I described several *incentive problems* relate to the findings in the literature: these problems relate to asymmetric information between the donor and the transfer recipient, private information within the donor community, and the strategic behavior of an individual resource country towards competitors for transfer payments and/or the donor community. Due to these incentive problems, an agreed upon arrangement may only lead to a level of conservation that is below a *Pareto-efficient* level from a global perspective. I supported the conceptual finding of a suboptimal outcome by summarizing empirical figures on currently provided international transfers and on estimates regarding the costs of a global network of protected areas. A comparison of the two sets of figures indicated that there is a substantial gap in the financing of protection that is (partly) attributable to a lack of international funding.

From the perspective of the transfer donor, *cost-effectiveness* depends upon the extent of conservation attained for a given amount of financial resources. Both the extent of conservation and the size of the transfer are determined in negotiations. When an international transfer is Pareto improving, this implies that a cooperation surplus is generated due to improved allocation of scarce resources. The agreement on conservation and the size of the transfer simultaneously determines the size of this surplus and its division among the stakeholders. The literature suggests that on the one hand, the cost-effectiveness of transfers determines the donor's willingness to pay and, on the other hand, the equitable sharing of the cooperation surplus influences the resource country's willingness to accept additional conservation that was originally not in its own interest.

I presented results of my own descriptive analysis of incremental cost assessments that have been carried out for GEF funded projects. These assessments describe how costs of extra conservation are shared by the international donor and the resource country. It turned out that, in practice, extra activities are either jointly financed or completely financed by the donor. I combined the quantitative empirical results with the conceptual findings on the cost-effectiveness of transfer and concluded that whenever extra activities generate an additional benefit for the resource country, complete financing by the GEF leaves a surplus equal to the additional benefit for the resource country.

I illustrated that complete financing by the donor can generally be consistent with a cost-minimizing arrangement of input in protected area management. However, if the resource country increasingly participates in the financing (according to the generated incremental domestic benefits), the donor can obtain the same level of conservation for a smaller transfer, i.e., from the donor's point of view the relative price of conservation decreases. When the price decreases, the donor has an incentive to use increasing funds as transfers to safeguard the conservation of other sites. As a result, the total level of conservation is improved relative to a situation in which extra activities are completed funded by international transfers.

Against this background, the results of my empirical analysis of GEF projects demonstrated that joint financing prevails in the majority of cases, i.e., the resource country participates in the financing of extra conservation. Although I could not ascertain the precise extent to which domestic benefits are taken into account, my findings suggest that the GEF mechanism generally recognizes the cost-effective use of transfer resources.

The agreed upon sharing of the costs of incremental conservation is the subject of negotiation, whereas resource countries may show varying degrees of *bargaining power* in negotiations with international donors. By using data from the GEF biodiversity project portfolio, I carried out a simple descriptive analysis to study whether the cost sharing of activities for the purpose of extra conservation is systematically biased towards relatively powerful resource countries. By classifying resource countries according to criteria that serve to indicate bargaining power, I studied the sharing of incremental expenditures. It turned out that cost sharing is not influenced significantly by the potential bargaining power of specific resource countries.

A further question with respect to efficiency is whether the current international transfer mechanisms are designed in a way that encourages resource countries to continue to reinforce their efforts in the domestic protected area policy. Transfers by the GEF are earmarked for incremental conservation efforts and provided on the basis of short-term or medium-term projects. In other words, the donors to the GEF do not commit to the long-term financing of the protected areas that generate ecosystem services of global importance. A resource country faces the problem of how to finance protection in excess of a domestically optimal level after external project funding has expired. In this respect, the purpose the international transfers serve also matters.

My empirical analysis of the GEF biodiversity projects illustrated that the need for *sustainable financing of protected areas* is recognized: the internationally funded projects often aim at the establishment of ecotourism facilities that assist the generation of future revenues to finance the recurrent tasks of protected area management. Furthermore, instead of providing a sequence of individual transfers, external funding is sometimes used to capitalize an environmental fund that provides regular disbursements for several protected areas. Whether financial sustainability and, thereby, the incentive for long-term conservation can be promoted in this way depends upon the circumstances that prevail at the sites considered, and the development of the market for tourism services and the global capital market. Both markets influence the revenues that can be reinvested in conservation. Evidence from the analysis of GEF biodiversity projects indicated that environmental funds are only addressed in a minority of projects.

To conclude, it remains unclear as to whether the project-led mechanism of international transfers is suitable for creating dynamic conservation incentives in the resource countries. Alternative regimes that provide a permanent long-term transfer flow to specific protected areas or specific resource countries have not been implemented.

Protected area policies lead to a globally suboptimal outcome if countries that benefit from these policies *free ride* in the financing of global biodiversity. Free riding leads to an undersupply of ecosystem services that display the character of a global public good. I related the issue of free riding to recent calls for the *financial strengthening of the GEF* as the most important financing institution for multilateral environmental assistance. It is interesting to observe whether the stakeholders in international biodiversity policy will heed these calls, i.e., whether there will be an increase in the GEF trust fund in the future through, for example, the extension of the donor community, increasing the contributions of existing donors, or a combination of both.

To study this question, I provided a conceptual and empirical analysis of the finance of the GEF. The findings suggested (1) that some newly industrialized and transition countries that have not joined the GEF replenishment process could come into question as additional donors in the future. Furthermore, (2) high income industrialized countries that carry a comparatively small burden in the finance of the GEF may be invited to increase their contributions beyond current levels. However, I also demonstrated that these implications may have to be qualified when differences in preferences for global environmental protection and the role of unilateral transfers as a substitute for transfers in a multilateral framework are taken into account. Although I presented some evidence that (potential) donors do not contribute according to the benefits they receive from global environmental protection, I demonstrated that free-riding behavior is difficult to detect.

By applying the findings in the literature on collective actions and noncooperative interactions to the financing of the GEF, I draw several conclusions: a future increase in contributions to the GEF trust fund may occur if the costs and benefits of multilateral coordination change in such a way that it is profitable for nondonor countries to join the financing of the GEF or, for donor countries, to increase their contribution in future replenishments. Without such a change in the costs and benefits, *moral suasion* does not lead to the same effect, since it can be expected that countries' decisions are motivated by their own national interests. Furthermore, given that other countries have a potential incentive to free ride, the contributions of an individual donor country may not (only) be guided by its own preference for environmental protection but, rather, by the aim to *sustain the current common arrangement* on the financing of the GEF as a second-best allocation.

The literature on collective action theory argues that in order to obtain a good match between the benefits received and the shared burden of collective actions, institutions of multilateral collaboration have to be rearranged. Since the restructuring of the GEF in 1994, fundamental *institutional changes* to this institution have not been discussed. In this regard, the findings of my study suggest that it can be worthwhile to analyze the interactions between developed and developing countries in the GEF replenishment process in more detail.

Furthermore, I interpreted the GEF as an institutional regime that represents an intermediate global public good that serves to produce the final global public good "environmental protection." The financing of this global public good competes with *other global public goods* for international funding. I discussed that the described shortfall of resources invested in the global network of protected areas may be due to the fact that the international community considers environmental protection to be of minor importance relative to other global public goods.

As they are aware that there is competition between global public goods, policymakers have established the principle of *additional funds* in the policies of the CBD and the GEF. This implies that the GEF donors are committed to providing resources for the conservation of the global environment in addition to the resources that have already been made available as official assistance for other purposes. It turned out that the empirical results of my analysis of GEF projects were not conclusive on this issue. Moreover, the results need to be extended to include information on unilateral and multilateral funding for the provision of other global public goods.

The issue of the *distributional effects* and political acceptance of international transfers and protected area policy again includes different levels of analysis. On a domestic level, the enforcement of a protected area policy depends upon the *participation of local communities* whose livelihoods are affected by protected area provisions. Political conflicts that arise from a lack of participation impede the enforcement of an effective protected area policy. Without going into detail, I described that integrated conservation and development projects are considered as a means to mitigate potential conflicts. In the literature, there are different opinions on whether conservation needs and local economic development can be reconciled or whether there is a noticeable trade-off between the two objectives.

On the international level, equity concerns relate to the described Pareto improvement induced by the provision of international transfers. As mentioned earlier, the *distribution of the cooperation surplus* generated from an improved allocation of resources is determined by the agreement on the size of transfer and the extent of extra conservation. Conceptual studies in this regard argue that resource countries, as transfer recipients, may refuse to cooperate with the donor community as long as the transfer offer does not provide them with an equitable/ larger share of the cooperation surplus. In the meantime, biodiversity is further degraded so that the attainable cooperation surplus shrinks. I pointed out that whether a resource country behaves strategically in this way depends upon certain conditions, such as a low domestic benefit from conserving its own resources, as well as irreversibility in biodiversity degradation. The provided empirical analysis of the GEF was not conclusive on the importance of strategic behavior, since only the agreed project activities were considered. This is because little information on failed project proposals is available.

Equity also matters in *burden sharing* among donor countries that cofinance the protection of globally important natural resources in the resource countries. Through the use of an empirical analysis, I identified disproportionality in the multilateral financing of the GEF, as well as in unilateral noncooperative actions. This disproportionality parallels that in the (multilateral) development financing described in the literature. Accordingly, disproportionality within the donor community is not inherent to the international financing of environmental protection, but is, rather, subject to general driving factors in international cooperation. This finding has to be kept in mind when making an effort to increase funds available for conservation.

The *administrative feasibility* of protected area policy and international transfers was outside in the focus of the study. However, with regard to protected area policy, the empirical analysis of GEF biodiversity projects showed that the transferred resources are used to improve protected area management rather than expand the current network of protected areas. In other words, the funds more frequently serve purchases of management input in protected areas already designated than of compensating forgone payoffs from land development. This result highlights the importance of the *transaction costs* of enforcing property rights to protected areas and the wildlife hosted therein; it is therefore an interesting finding for future studies of protected area development.

References

Achard, F., H.D. Eva, H.-J. Stibig, P. Mayaux, J. Gallego, T. Richards, and J.-P. Malingreau (2002). Determination of deforestation rates of the world's humid tropical forests. *Science* 297: 999–1002.

Acharya, R. (1991). Patenting biotechnology. GATT and the erosion of world's biodiversity. *Journal of World Trade* 25(6): 71–87.

Addison, T., M. McGillivray, and M. Odedokun (2004). Donor funding of multilateral aid agencies: Determining factors of revealed burden sharing. *The World Economy* 27(2): 173–191.

Aghion, P., and P. Howitt (1992). A model of growth through creative destruction. *Econometrica* 60(2): 323–351.

Alchian, A.A. (1965). Some Economics of Property Rights. *Il Politico* 30(4): 816–829.

Allem, A.C. (2000). The terms genetic resources, biological resource, and biodiversity examined. *Environmentalist* 20(4): 335–341.

Amelung, T. (1991). Internationale Transferzahlungen zur Lösung globaler Umweltprobleme dargestellt am Beispiel der tropischen Regenwälder. *Zeitschrift für Umweltpolitik und Umweltrecht* 14(2): 159–178.

Amelung, T. (1993). Tropical deforestation as an international economic problem. In H. Giersch (ed.), *Economic progress and environmental concerns*. Berlin: Springer.

Anand, P.B. (2004). Financing the provision of global public goods. *The World Economy* 27(2): 215–237.

Ando, A., J. Camm, S. Polasky, and A. Solow (1998). Species distribution, land values and efficient conservation. *Science* 279: 2126–2128.

Andreoni, J. (1988). Privately provided public goods in a large economy: The limits of altruism. *Journal of Public Economics* 35(1): 57–73.

Angelsen, A., and D. Kaimowitz (1999). Rethinking the causes of deforestation: Lessons from economic models. *The World Bank Research Observer* 14(1): 73–98.

Armsworth, P.R., B.E. Kendall, and F.W. Davis (2004). An introduction to biodiversity concepts for environmental economists. *Resource and Energy Economics* 26(2): 115–136.

Arrow, K., and A.C. Fisher (1974). Environmental preservation, uncertainty, and irreversibility. *Quarterly Journal of Economics* 88(2): 312–319.

Arrow, K., B. Bolin, R. Costanza, P. Dasgupta, C. Folke, C.S. Holling, B. Jansson, S. Levin, K. Mäler, C. Perrings, and D. Pimentel (1995). Economic growth, carrying capacity, and the environment. *Science* 268: 520–521.

Artuso, A. (1996a). Economic analysis of biodiversity as a source of pharmaceuticals. In PAHO (Pan American Health Organization) (ed.), *Biodiversity, biotechnology, and sustainable development in health and agriculture: Emerging connections*. Washington, D.C.

Artuso, A. (1996b). Economic analysis of contractual arrangements for biochemical prospecting. PAHO (Pan American Health Organization) (ed.), *Biodiversity, biotechnology, and sustainable development in health and agriculture: Emerging connections*. Washington, D.C.

Artuso, A. (1997). Natural product research and emerging market for biochemical resources. *Journal of Research in Pharmaceutical Economics* 8(2): 3–23.

Artuso, A. (2002). Bioprospecting, benefit sharing and biotechnological capacity building. *World Development* 30(8): 1355–1368.

Asafu-Adjaye, J. (2003). Biodiversity loss and economic growth: A cross-country analysis. *Contemporary Economic Policy* 21(2): 173–185.

ASSINSEL (International Association of Plant Breeders for the Protection of Plant Varieties) (1996). Position on maintenance of and access to plant genetic resources for food and agriculture (PGRFA). Available at <http://www.worldseed.org/positions.htm>, accessed on May 15, 2005.

ASTA (American Seed Trade Association) (2005). Seed statistics. Available at <http://www.amseed.com/about_statistics.asp>, accessed on December 30, 2005.

Aylward, B.A. (1993). The economic value of pharmaceutical prospecting and its role in biodiversity conservation. LEEC Paper DP 93-05. International Institute for Environment and Development, London Environmental Economic Centre, London.

Aylward, B.A. (1995). The role of plant screening and plant supply in biodiversity conservation, drug development and health care. In T. Swanson (ed.), *Intellectual property rights and biodiversity conservation: An interdisciplinary analysis of the values of medicinal plants*. Cambridge: Cambridge University Press.

Baillie, J.E.M., C. Hilton-Taylor, and S.N. Stuart (eds.) (2004). 2004 IUCN Red List of Threatened Species. A Global Species Assessment. Gland: IUCN.

Balland, J.-M., and J.-P. Platteau (1997). Wealth inequality and efficiency in the commons. Part I: The unregulated case. *Oxford Economic Papers* 49(4): 451–482.

Balmford, A. (2003). The global costs and benefits of conserving wild nature. Paper presented at the IUCN 5th World Parks Congress: Sustainable Finance Stream, September 8–17, Durban.

Balmford, A., A. Bruner, P. Cooper, R. Costanza, S. Farber, R.E. Green, M. Jenkins, P. Jefferiss, V. Jessamy, J. Madden, K. Munro, N. Myers, S. Naeem, J. Paavola, M. Rayment, S. Rosendo, J. Roughgarden, Kate Trumper, and R.K. Turner (2002). Economic reasons for conserving wild nature. *Science* 297: 950–953.

Barbault, R., and S.D. Sastrapradja (1995). Generation, maintenance and loss of biodiversity (chapter 4). In V.H. Heywood (ed.), *Global biodiversity assessment. United Nations Environment Programme*. Cambridge: Cambridge University Press.

Barbier, E.B. (1997). Ecological economics, uncertainty and implications for policy setting priorities in biodiversity conservation. In OECD (ed.), Investing in biological diversity: The Cairns Conference. Proceedings of the OECD International Conference on Incentive Measures for the Conservation and the Sustainable Use of Biological Diversity, Cairns, 25–28 March 1996. Paris.

Barbier, E.B., and B.A. Aylward (1996). Capturing the pharmaceutical value of biodiversity in a developing country. *Environmental and Resource Economics* 8(2): 157–181.

Barbier, E.B., and E.H. Bulte (2004). Introduction to the symposium on trade, renewable resources and biodiversity. *Journal of Environmental Economics and Management* 48(2): 883–890.

Barbier, E.B., and J.C. Burgess (1997). The economics of tropical forest land use options. *Land Economics* 73(2): 174–195.

Barbier, E.B., and J.C. Burgess (2001). The economics of tropical deforestation. *Journal of Economic Surveys* 15(3): 413–433.

Barbier, E.B., and M. Rauscher (1994). Trade, tropical deforestation and policy intervention. *Environmental and Resource Economics* 4(1): 75–90.

Barbier, E.B., and C.-E. Schulz (1997). Wildlife, biodiversity and trade. *Environment and Development Economics* 2(2): 145–172.

Barbier, E.B., and I. Strand (1998). Valuing mangrove-fishery linkages: A case study of Campeche, Mexico. *Environmental and Resource Economics* 12(2): 151–166.

Barrett, S. (1992). Economic growth and environmental preservation. *Journal of Environmental Economics and Management* 23(3): 289–300.

Barrett, S. (1993). Optimal economic growth and the conservation of biological diversity. In E.B. Barbier (ed.), *Economics and ecology: New frontiers and sustainable development.* London: Chapman & Hall.

Barrett, S. (1994a). The biodiversity supergame. *Environmental and Resource Economics* 4(2): 111–122.

Barrett, S. (1994b). Self-enforcing international environmental agreements. *Oxford Economic Papers* 46(Supplement): 878–894.

Barrett, S. (1996). Economic incentives for the protection of biological diversity. In H. Siebert (ed.), *Elemente einer rationalen Umweltpolitik.* Expertisen zur umweltpolitischen Neuorientierung. Tübingen: Mohr Siebeck.

Barrett, S. (2001). International cooperation for sale. *European Economic Review* 45(10): 1835–1850.

Barrett, S. (2002). Supplying international public goods. How nations cooperate. In M. Ferroni and A. Mody (eds.), *International public goods: Incentives, measurement, and financing.* Washington, D.C.: World Bank.

Barrett, S. (2003). *Environment and statecraft: The strategy of environmental treaty-making.* Oxford: Oxford University Press.

Barton, J.H., and W.E. Siebeck (1994). Material transfer agreements in genetic resources exchange: The case of the International Agricultural Research Centres. Issues in Genetic Resources No. 1, May 1994. Rome: International Plant Genetic Resources Institute.

Batabyal, A.A. (2000). Aspects of ecosystem persistence and the optimal conservation of species. *International Review of Economics and Finance* 9(1): 69–77.

Batabyal, A.A. (2002). Human actions, the survival of keystone species, and the resilience of ecological-economic systems. *Resources Policy* 28(3/4): 153–157.

Batabyal, A.A., J.R. Kahn, and R.V. O'Neill (2003). On the scarcity value of ecosystem services. *Journal of Environmental Economics and Management* 46(2): 334–352.

Baumgärtner, S. (2002). Measuring the diversity of what? And for what purpose? A conceptual comparison of ecological and economic measures of biodiversity. Mimeo. Interdisciplinary Institute for Environmental Economics, University of Heidelberg, Heidelberg.

Baumol, W.J., and W.E. Oates (1988). *The theory of environmental policy*. 2nd edition. Cambridge: Cambridge University Press.

Bellon, M.R. (2004). Conceptualizing interventions to support on-farm genetic resources conservation. *World Development* 32(1): 159–172.

Bergin, J., and W.B. MacLeod (1993). Efficiency and renegotiation in repeated games. *Journal of Economic Theory* 61(1): 42–73.

Bergstrom, T.C., L. Blume, and H. Varian (1986). On the private provision of public goods. *Journal of Public Economics* 29(1): 25–49.

Bernheim, B.D. (1986). On the voluntary and involuntary provision of public goods. *The American Economic Review* 76(4): 789–793.

Besen, S.M., and L.J. Raskind (1991). An introduction to the law and economics of intellectual property. *Journal of Economic Perspectives* 5(1): 3–27.

Bhat, M.G. (1996). Trade-related intellectual property rights to biological resources: Socioeconomic implications for developing countries. *Ecological Economics* 19(3): 205–217.

Bhat, M.G. (1999). On biodiversity, intellectual property rights, and conservation. *Ecological Economics* 29(3): 391–403.

Bishop, R.C. (1978). Endangered species and uncertainty: The economics of a safe minimum standard. *American Journal of Agricultural Economics* 60(1): 10–18.

Blandford, D., and R.N. Boisvert (2002). Non-trade concerns and domestic/international policy choice. Working Paper #02-1. International Agricultural Trade Research Consortium (IATRC), Pullman, Washington.

Boadway, R., and M. Hayashi (1999). Country size and the voluntary provision of international public goods. *European Journal of Political Economy* 15(4): 619–638.

Boisson de Chazournes, L. (2003). The Global Environmental Facility as a pioneering institution. Working Paper 19. The Global Environmental Facility, Washington, D.C.

Boyer, M. (1988). Burden-sharing and comparative advantage in the Western alliance system: Toward a new economic theory of international cooperation. Dissertation. University of Maryland.

Boyer, M. (1990). A simple and untraditional analysis of Western alliance burden-sharing. *Defense Economics* 1: 243–259.

Bragdon, S.H. (2003). The multilateral system of genetic resources exchange: Why trade in food genetic resources matters? *Bridges Monthly Review* (International Centre for Trade and Sustainable Development (ICTSD)) 7(6): 20–22.

Bragdon, S.H., and D.R. Downes (1998). Recent policy trends and developments related to the conservation, use and development of genetic resources. Issues in Genetic Resources No. 7, June 1998. Rome: International Plant Genetic Resources Institute.

Brander, J.A., and M.S. Taylor (1998). Open access renewable resources: Trade and trade policy in a two-country model. *Journal of International Economics* 44(2): 181–373.

Brandon, K. (1996). Ecotourism and conservation: A review of key issues. Environmental Department Papers No.033. Global Environmental Division, World Bank, Washington, D.C.

Brandon, K., and M. Wells (1992). Planning for people and parks: Design dilemmas. *World Development* 20(4): 557–570.

Brock, W.A., and A. Xepapadeas (2001). Valuing biodiversity from an economic perspective: A unified economic, ecological, and genetic approach. *The American Economic Review* 93(5): 1597–1614.

Brock, W.A., and A. Xepapadeas (2002). Optimal ecosystem management when species compete for limiting resources. *Journal of Environmental Economics and Management* 44(2): 189–220.

Brown, G.M., and J.F. Shogren (1998). Economics of the Endangered Species Act. *The Journal of Economic Perspectives* 12(3): 3–20.

Bruce, N. (1990). Defence expenditures by countries in allied and adversarial relationships. *Defense Economics* 1: 179–195.

Bruner, A. (2003). How much will effective protected area system cost? Center for Applied Biodiversity Science, Conservation International, Washington, D.C.

Brush, S.B. (1996). Valuing crop genetic resources. *Journal of Environment & Development* 5(4): 416–433.

Brush, S.B. (2002). The lighthouse and the potato: Internalizing the value of crop genetic diversity. Working Paper Series No.37. Political Economy Research Institute, University of Massachusetts, Amherst.

Brush, S.B., E. Taylor, and M. Bellon (1992). Technology adoption and biological diversity in Andean potato agriculture. *Journal of Development Economics* 39(2): 365–387.

Bryson, N.S., and C.E. Kaczmarek (2000). Cradas: The national parks capitalize on their genetic resources. *Environmental Quality Management* 10(2): 93–100.

Buchanan, J.M. (1966). Joint supply, externality and optimality. *Economia* 33(132): 404–415.

Buchholz, W., and C. Haslbeck (1991). Private Verhandlungen und staatliche Regulierung bei asymmetrischer Information. *Finanzarchiv* 49(2): 167–180.

Bulte, E.H., and E.B. Barbier (2005). Trade and renewable resources in a second best world: An overview. *Environmental and Resource Economics* 30(4): 423–463.

Bulte, E.H., and G.C. van Kooten (1999). Economic efficiency, resource conservation and the ivory trade. *Ecological Economics* 28(2): 171–181.

Bulte, E.H., R.D. Horan, and J.F. Shogren (2003). Elephants: Comment. *The American Economic Review* 93(4): 1437–1445.

Burgess, J.C. (1993). Economics of the trade in endangered species: Implications for sustainable management. In H. Giersch (ed.), *Economic progress and environmental concerns*. Berlin: Springer.

Butler, D. (1998). World bank calls for a fairer deal on patents and knowledge. *Science* 395: 529.

Byström, M., P. Einarsson, and G. Axelsson Nycander (1999). Fair and equitable: Sharing the benefits from use of genetic resources and traditional knowledge. Stockholm: Swedish Scientific Council on Biological Diversity.

Campell, J.G., and A. Martin (2000): Financing the global benefits of forests: The Bank's GEF portfolio and the 1991 Forest Strategy: A review of the World Bank's 1991 forest strategy and its implementation. World Bank Operations Evaluation Department, The World Bank, Washington D.C.

Carraro, C., and D. Siniscalco (1993). Strategies for the international protection of the environment. *Journal of Public Economics* 52(3): 309–328.

Carraro, C., and D. Siniscalco (1998). International environmental agreements: Incentives and political economy. *European Economic Review* 42(3): 561–572.

Carraro, C., and D. Siniscalco (2001). Transfers, commitments. And issue linkage in international environmental negotiations. In A. Ulph (ed.), *Environmental policy, international agreements, and international trade*. Oxford: Oxford University Press.

Carson, R.T., L. Wilks, and D. Imber (1994). Valuing the preservation of Australia's Kakadu Conservation Zone. *Oxford Economic Papers* 46(Supplement): 727–749.

Cartier, C.M., and H.J. Ruitenbeek (1999). Review of the biodiversity valuation literature. In H.J. Ruitenbeek and C.M. Cartier (eds.), *Issues in applied coral reef biodiversity valuation: Results for Montego Bay, Jamaica*. World Bank Research Committee Project RPO#682-22 Final Report, The World Bank, Washington, D.C.

Cervigni, R. (1998). Incremental cost in the Convention on Biological Diversity. *Environmental and Resource Economics* 11(2): 217–241.

Cervigni, R. (2001). *Biodiversity in the balance: Land use, national development and global welfare*. Cheltenham: Elgar.

Cervigni, R., and D. Pearce (1995). North-south resource transfers, incremental cost and the Rio Conventions. CSERGE Working Paper No. GEC 95-33. CSERGE, Norwich.

Chander, P., and H. Tulkens (1997). The core of an economy with multilateral environmental externalities. *International Journal of Game Theory* 26(3): 379–401.

Chape, S., S. Blyth, L. Fish, P. Fox, and M. Spalding (compilers) (2003): 2003 United Nations list of protected areas. Gland: IUCN, Gland and Cambridge, UK: UNEP-WCMC.

Chapin, F.S. III, E.S. Zavaleta, V.T. Eviner, R.L. Naylor, P.M. Vitousek, H.L. Reynolds, D.U. Hooper, S. Lavorel, O.E. Sala, S.H. Hobbie, M.C. Mack, and S. Diaz (2000). Consequences of changing biodiversity. *Nature* 405: 234–242

Chavas, J.-P. (2000). Ecosystem valuation under uncertainty and irreversibility. *Ecosystems* 3(1): 11–15.

Chen, L.C., T.G. Evans, and R.A. Cash (1999). Health as a global public good. In I. Kaul et al. (eds.), *Global public goods: International cooperation in the 21st century.* New York: Oxford University Press.

Chichilnisky, G. (1994). North-South trade and the global environment. *The American Economic Review* 84(4): 851–874.

Chiesura, A., and R. de Groot (2003). Critical natural capital: A socio-cultural perspective. *Ecological Economics* 44(2–3): 219–231.

Chomitz, K.M., and K. Kumari (1996). The domestic benefits of tropical forests: A critical review. CSERGE Working Paper No. GEC 96-19. CSERGE, Norwich.

Chomitz, K.M., K. Alger, T.S. Thomas, H. Orlando, and P.V. Nova (2005). Opportunity costs of conservation in a biodiversity hotspot: The case of southern Bahia. *Environment and Development Economics* 10(3): 293–312.

Chou, C.-F., and O. Shy (1993). The crowding-out effects of long duration of patents. *The Rand Journal of Economics* 24(2): 304–312.

Ciriacy-Wantrup, S.V. (1952). *Resource conservation: Economics and policies.* Berkeley: University of California Press.

Clemençon, R. (2000). Financing protection of the global commons. Working Paper 17. Global Environmental Facility (GEF), Washington, D.C.

Coase, R.H. (1937). The nature of the firm. *Economia* 4(4): 386–405.

Coase, R.H. (1960). The problem of social choice. *Journal of Law and Economics* 4(1): 1–44.

Cole, D., and P. du Plessis (2001). Nambian devil's claw. A case study on benefit-sharing arrangements. Centre for Research Information Action in Africa, Southern African Development Consulting (CRIAA-SA-DC), Windhoek.

Connolly, B. (1996). Increments for the Earth: The politics of environmental Aid. In R.O. Keohane and M.A. Levy (eds.), *Institutions for environmental aid: Pitfalls and promise.* Cambridge, Mass.: MIT Press.

Conover, W.J. (1971). *Practical non-parametric statistics.* New York: John Wiley & Sons.

Cooper, D., J. Engels, and E. Frison (1994). A multilateral system for plant genetic resources: Imperatives, achievements and challenges. Issues in Genetic Resources No. 2, May 1994. Rome: International Plant Genetic Resources Institute.

Cooper, J.C. (2001). Construction of a fund for sharing the benefits from the utilization of plant genetic resources for food and agriculture. *Environment and Development Economics* 6(1): 47–62.

Coram, A.T. (2003). Maintaining environmental resources in the international system: A comparison of contractual arrangements. *Social Science Quarterly* 84(1): 174–190.

Cornes, R., and T. Sandler (1984). Easy riders, joint production, and public goods. *Economic Journal* 94(375): 570–598.

Cornes, R., and T. Sandler (1985). The simple analytics of pure public good provision. *Economica* 52(205): 103–116.

Cornes, R., and T. Sandler (1994). The comparative static properties of the impure public good model. *Journal of Public Economics* 54(3): 403–421

Cornes, R., and T. Sandler (1996). *The theory of externalities, public goods, and club goods.* 2nd Edition. Cambridge: Cambridge University Press.

Cornes, R., R. Hartley, and T. Sandler (1999). Equilibrium existence and uniqueness in public good models: An elementary proof via contradiction. *Journal of Public Economic Theory* 1(4): 499–509.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, and J. Paruelo (1997). The value of the world's ecosystem services and natural capital. *Nature 387: 253–260?*

Costello, C., and C. McAusland (2003). Protectionism, trade and measures from exotic species introductions. *American Journal of Agricultural Economics* 85(4): 964–975.

Costello, C., and S. Polasky (2004). Dynamic reserve site selection. *Resource and Energy Economics* 26(2): 157–174.

Costello, C., and M. Ward (2003). Search, bioprospecting, and biodiversity conservation: Comment. Working Paper. University of California, Santa Barbara.

Craft, A., and R.D. Simpson (2001). The value of biodiversity in pharmaceutical research with differentiated products. *Environmental and Resource Economics* 18(1): 1–17.

Cragg, G., D. Newman, and K. Snader. (1997). Natural products in drug discovery and development. *Journal of Natural Products* 60(1): 52–60.

Crist, T.C. (2003). The spatial structure of biodiversity across landscape and regions. In V. Wolters et al. Biodiversity in the cultural landscape. An international workshop of BIOLOG Europe. *Workshop Report.* Giessen (Schloss Rauischholzhausen).

Crowards, T.M. (1998). Safe minimum standards: Costs and opportunities. *Ecological Economics* 25(3): 303–314

Cullen, R. (1994). Antarctic minerals and conservation. *Ecological Economics* 10(2): 143–155.

Cunningham, A.B. (1993). African medicinal plants: Setting priorities at the interface between conservation and primary health care. People and Plants working paper 1. Paris. UNESCO. Available at <http://peopleandplants.org/whatweproduce/Working papers/pdf/wp1e.pdf>, accessed on December 30, 2005.

Daily, G. (1997). *Nature's services: Societal dependence on natural ecosystems.* Washington, D.C: Island Press.

Dasgupta, P., and K.-G. Mäler (2003). The economics of non-convex ecosystems: Introduction. *Environmental and Resource Economics* 26(4): 499–525.

Day, K.A., and G.B. Frisvold (1993). Medical research and genetic resources management: The case of Taxol. *Contemporary Policy Issues* 11(3): 1–11.

de Groot, R.S., M.A. Wilson, and R.M.J. Boumans (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics* 41(3): 393–408.

Deacon, R.T., and P. Murphy (1997). The structure of environmental transaction: The debt-for-nature swap. *Land Economics* 73(1): 1–24.

Dedeurwaerdere, T. (2005). From bioprospecting to reflexive governance. *Ecological Economics* 53(4): 473–491.

Demsetz, H. (1967). Towards a theory of property rights. *The American Economic Review* 57(2): 347–359.

Dhillion, S.S., H. Svarstad, C. Amundsen, and H.C. Bugge (2002). Bioprospecting: Effects on environmnent and development. *Ambio* 31(6): 491–493.

Di Falco, S., and C. Perrings (2003). Crop Genetic Diversity, Productivity and Stability of Agroecosystems. A Theoretical and Empirical Investigation. *Scottish Journal of Political Economy* 50(2): 207–216.

Di Masi, J.A., R.W. Hansen, and H.G. Grabowski (2003). The price of innovation: New estimates of drug development costs. *Journal of Health Economics* 22: 151–185.

Dietz, S., and W.N. Adger (2003). Economic growth, biodiversity loss and conservation effort. Centre for Social and Economic Research on the Global Environment. CSERGE Working Paper No. ECM 01-03. CSERGE, Norwich.

Downes, D.R. (2002). How intellectual property could be a tool to protect traditional indigenous knowledge. In K.P. Gallagher and J. Werksman (eds.), *The Earthscan reader on international trade and sustainable development.* London: Earthscan.

Downes, D.R., and S.A. Laird (1999). Community registries of biodiversity-related knowledge. The role of intellectual property in managing access and benefit sharing. Paper Prepared for UNCTAD Biotrade Initiative.

Droege, S., and B. Soete (2001). Trade-related intellectual property rights, North-South trade, and biological diversity. *Environmental and Resource Economics* 19(2): 149–163.

Dudley, L. (1979). Foreign aid and theory of alliances. *The Review of Economics and Statistics* 61(4): 564–571.

Dunleavy, K.J., and M.W. Vinnola (2000). A comparative review of patenting of biotechnological inventions in the United States and Europe. *The Journal of World Intellectual Property* 3(1): 65–76.

Dutfield, G. (1999). Intellectual property rights, trade and biodiversity: The case of seeds and plant varieties. Background paper. Intersessional meeting on the operations of the Convention, Montreal 28–30 June 1999. Gland: IUCN.

Dutfield, G. (2000a). Biodiversity in industrial research and development: Implications for developing countries. *International Journal of Biotechnology* 2(1–3): 103–114.

Dutfield, G. (2000b). The public and private domains: Intellectual property rights in traditional knowledge. *Science Communication* 21(3): 274–295.

Dutfield, G. (2000c). Sharing the benefits of biodiversity: Is there a role for the patent system? *The Journal of World Intellectual Property* 5(6): 899–931.

EC/EDG (European Commission, Environment Directorate-General) (2003). LIFE for Natura 2000: 10 years implementing the regulation. Luxembourg.

EC-CHM (European Community Clearing House Mechanism) (2005). EC Biodiversity Clearing House Mechanism. Available at <http://biodiversity-chm.eea.eu.int/nyglossary_terms>, accessed on December, 30 2005.

Economist (2003a). Reinvesting yesterday. Biotech's biggest use may be to rebuild basic industries. 366(8317): 14–16.s

Economist (2003b). There be dragons—Komodo and Manado. New ways of protecting wildlife. 16–22 October (8346): 58.

ECPA (European Crop Protection Association) (2004). Take a closer look. ECPA Annual Report 2002/2003. Brussels.

Eggertsson, T. (1990). *Economic behavior and institutions.* Cambridge: Cambridge University Press.

Eiswerth, M.E., and J.C. Haney (2001). Maximizing conserved biodiversity: Why ecosystem indicators and thresholds matter. *Ecological Economics* 38(2): 259–274.

Ekins, P., C. Folke, and R. de Groot (2003). Identifying critical natural capital. *Ecological Economics* 44(2–3): 159–163.

Endres, A. (1995). Zur Ökonomie internationaler Umweltvereinbarungen. *Zeitschrift für Umweltpolitik* 18(2): 143–178.

Endres, A., and V. Radke (1999). Land use, biodiversity, and sustainability. *Journal of Economics* 70(1): 1–16.

Evenson, R.E. (1996). Economic valuation of biodiversity for agriculture. PAHO (Pan American Health Organization) (ed.), *Biodiversity, biotechnology, and sustainable development in health and agriculture: Emerging connections.* Washington, D.C.

Evenson, R.E., and D. Gollin (1997). Genetic resources, international organizations, and improvement in rice varieties. *Economic Development and Cultural Change* 45(3): 471–500.

Evenson, R.E., and Y. Kislev (1976). A stochastic model of applied research. *Journal of Political Economy* 84(2): 265–282.

Fairman, D. (1996). The Global Environment Facility: Haunted by the shadow of the future. In R.O. Keohane and M.A. Levy (eds.), *Institutions for environmental aid: Pitfalls and promise*. Cambridge, Mass.: MIT Press.

Falck-Zepeda, J.B., G. Traxler, and R.G. Nelson (2000). Surplus distribution from the introduction of a biotechnology innovation. *American Journal of Agricultural Economics* 82(2): 360–369.

FAO (Food and Agriculture Organization of the United Nations) (1993). International Code of Conduct for Plant Germplasm Collecting and Transfer. Paper CPGR/93/8, Commission on Plant Genetic Resources, Fifth Session, Food and Agriculture Organization of the United Nations, Rome.

FAO (Food and Agriculture Organization of the United Nations) (1998). The state of the world's plant genetic resources for food and agriculture. Food and Agriculture Organization of the United Nations, Rome.

Farber, S.C., R. Costanza, and M.A. Wilson (2002). Economic and ecological concepts for valuing ecosystem services. *Ecological Economics* 41(3): 375–392.

Farnsworth, N.R., and R.W. Morris (1976). Higher plants: The sleeping giant of drug development. *American Journal of Pharmacology* 148: 46–52.

Farnsworth, N.R., and D.D. Soejarto (1985). Potential consequence of plant extinction in the United States on the current and future availability of prescription drugs. *Economic Botany* 39(3): 231–40.

Farrell, J. (1987). Information and the Coase Theorem. *Journal of Economic Perspectives* 1(2): 113–129.

Farrell, J., and E. Maskin (1989). Renegotiation in repeated games. *Games and Economic Behavior* 1(4): 327–360.

Ferraro, P.J. (2004). Targeting conservation investments in heterogeneous landscapes: A distance-function approach and application to watershed management. *American Journal of Agricultural Economics* 86(4): 905–918.

Ferraro, P.J., and A. Kiss (2002). Direct payments to conserve biodiversity. *Science* 298: 1718–1719.

Ferraro, P.J., and R.D. Simpson (2002). The cost-effectiveness of conservation payments. *Land Economics* 78(3): 339–353.

Ferroni, M., and A. Mody (2002). Global incentives for international public goods. Introduction and Overview. In M. Ferroni and A. Mody (eds.), *International public goods: Incentives, measurement, and financing*. World Bank. Washington, D.C.

FES (Friedrich-Ebert-Stiftung) (1997). Nord-Süd-Biopolitik—Zur Frage gentechnischer Vorherrschaft. Wieviel Modellhaftes bietet INBio/Costa Rica? Diskussion Runder Tisch II (Naumann). Available at <http://www.fes.de/interntl/umwelt/nord_sued/gis_1.html#12>, accessed on December 30, 2005.

Finus, M. (2004). International cooperation to resolve international pollution problems. Nota die Lavoro CCMP 41.2004. Fondazione Eni Enrico Mattei, Milan.

Finus, M., and B. Rundshagen (2001). Endogenous coalition formation in global pollution control. Nota di Lavoro 43.2001. Fondazione Eni Enrico Mattei, Milan.

Fisher, A.C., and W.M. Hanemann (1986). Option value and the extinction of species. *Advances in Applied Microeconomics* 4: 168–190.

Fisher, A.C., J.V. Krutilla, and C.J. Cicchetti (1972). The economics of environmental preservation: A theoretical and empirical analysis. *The American Economic Review* 62(4): 605–619.

Folke, C., C.S. Holling, and C. Perrings (1996). Biological diversity, ecosystems, and the human scale. *Ecological Applications* 6(4): 1018–1024.

Folke, C., S. Carpenter, T. Elmqvist, L. Gunderson, C.S. Holling, B. Walker, J. Bengtsson, F. Berkes, J. Colding, K. Danell, M. Falkenmark, L. Gordon, R. Kasperson, N. Kautsky, A. Kinzig, S. Levin, K. Maler, F. Moberg, L. Ohlsson, P. Olsson, E. Ostrom, W. Reid, J. Rockström, H. Savenije, and U. Svedin (2002). Resilience and sustainable development: Building adaptive capacity in a world of transformations. Scientific background paper on resilience for the process of The World Summit on Sustainable Development on behalf of The Environmental Advisory Council to the Swedish Government. Stockholm, Sweden.

Fowler, C. (2000). Establishing the scope for a multilateral system for Plant Genetic Resources for Food and Agriculture: Implications of crop exclusions. *Biopolicy* 3(2).

Fowler, C., M. Smale, and S. Gaiji (2000). Germplasm flows between developing countries and the CGIAR: An initial assessment. Global Forum On Agricultural Research (GFAR). Rome: GFAR Secretariat/FAO.

Fowler, C., M. Smale, and S. Gaiji (2001). Unequal exchange? Recent transfer of agricultural resources and their implications for developing countries. *Development Policy Review* 19(2): 181–204.

Frisvold, G.B., and P.T Condon (1998). The Convention on Biological Diversity and Agriculture: Implications and unresolved debates. *World Development* 26(4): 551–570.

Frisvold, G.B., and P. Cordon (1994). Biodiversity conservation and biotechnology development agreements. *Contemporary Policy* 12(3): 1–9.

Fromm, O. (2000). Ecological structure and functions of biodiversity as elements of its total economic value. *Environmental and Resource Economics* 16(3): 303–328.

Furubotn, E.G., and S. Pejovich (1972). Property rights and economic theory: A survey of recent literature. *Journal of Economic Literature* 10(4): 1137–1162.

Furubotn, E.G., and S. Pejovich (1974). *The Economics of Property Rights.* Cambridge, Mass.: Ballinger Publications.

Gadgil, M. (1996). Patenting life. *Artha vijnana /Journal of the Gokhale Institute of Politics and Economics* 37(4): 307–321.

GAIA/GRAIN (Genetic Resources Action International) (2000). Biodiversity for sale: Dismantling the hype about benefit sharing. Global trade and biodiversity in conflict. Issue No.4. Barcelona.

Gale, F.P. (2000). Economic specialization versus ecological diversification: The trade policy implications of taking the ecosystem approach seriously. *Ecological Economics* 34(3): 285–292.

Gaston, K.J. (2000). Global patterns in biodiversity. *Science* 405: 220–227.

GEF (Global Environment Facility) (1996). Incremental costs. GEF/C.7/Inf.5. Washington, D.C.

GEF (Global Environmental Facility) (1998a). Experience with Conservation Trust Funds. GEF/C.12/Inf.6. Washington, D.C.

GEF (Global Environmental Facility) (1998b). Funds and Trust funds. GEF/C.12/Inf.5. Washington D.C.

GEF (Global Environment Facility) (1998c). Progress on incremental costs. GEF/C.12/Inf.4. Washington, D.C.

GEF (Global Environment Facility) (2000). GEF project cycle. GEF/C.16/Inf.7. October 5, 2002. Geneva.

GEF (Global Environment Facility) (2001a). Burden sharing for the third GEF replenishment. GEF/R.3/7/Rev.1. April 26, 2001. Washington, D.C.

GEF (Global Environment Facility) (2001b). GEF-2 pledges and funding situation. GEF/R.3/4/Rev.1. May 4, 2001. Washington, D.C.

GEF (Global Environment Facility) (2001c). GEF-2 commitments authority and current funding status. GEF/R.3/12. September 20, 2001. Washington, D.C.

GEF (Global Environment Facility) (2001d). Potential reference currencies for use in the third replenishment. GEF/R.3/13. September 20, 2001. Washington, D.C.

GEF (Global Environment Facility) (2001e). Second overall performance study of the GEF: Note on preparation of policy recommendations. GEF/R.3/20. November 20, 2001. Washington, D.C.

GEF (Global Environment Facility) (2001f). Burden sharing and financial issues for the third GEF replenishment. GEF/R.3/21. November 20, 2001. Washington, D.C.

GEF (Global Environment Facility) (2002a). Programming of resources for the third GEF replenishment (Revised tabulation). GEF/R.3/15/Add.1 February 11, 2002. Washington, D.C.

GEF (Global Environment Facility) (2002b). Procurement restrictions related to arrears in donors' contribution to the GEF. GEF/R.3/25. February 13, 2002. Washington, D.C.

GEF (Global Environment Facility) (2002c). GEF-2 current and projected funding status. GEF/R.3/35. July 29, 2002. Washington, D.C.

GEF (Global Environment Facility) (2002d). Draft summary of negotiations. GEF/R.3/38. August 1, 2002. Washington, D.C.

GEF (Global Environment Facility) (2002e). Summary of negotiations on the third GEF replenishment. November 5, 2002. Washington, D.C.

GEF (Global Environment Facility) (2003). Trustee report on the financial status and management of the GEF trust fund. GEF/C.22/Inf.3. October 29, 2003. Washington, D.C.

GEF (Global Environment Facility) (2004a). GEF and the Convention on Biological Diversity. A strong partnership with solid results. Washington, D.C.

GEF (Global Environment Facility) (2004b). Instrument for the establishment of the re-constructed Global Environment Facility. Reprint, first Printing 1994. Washington, D.C.

GEF (Global Environment Facility) (2005a): Project database. Available at <http://www.gefonline.org/home.cfm>, accessed on December 30, 2005.

GEF (Global Environment Facility) (2005b): Focal areas. Available at <http://gefweb.org/Projects/focal_areas/focal_areas.html >, on December 30, 2005.

GEF (Global Environment Facility) (2005c): Operational strategy of the Global Environment Facility. Section 2: Biodiversity. Available at <http://www.gefweb.org/public/opstrat/ch2.htm>, accessed on December 30, 2005.

GEF (Global Environment Facility) (2005d). Replenishment. Available at <http://gefweb.org/Replenishment/replenishment.html>, accessed on December 30, 2005.

Geist, H.J, and E.F. Lambin (2002). Proximate causes and underlying driving forces of tropical deforestation. *Bioscience* 52(2): 143–150.

Ginannakas, K. (2003). Infringement of intellectual property rights: Developing countries, agricultural biotechnology, and the TRIPs agreement. Brief 5:1-4. Research at a glance: Biotechnology and Genetic Resource Policies. IFPRI (International Food Policy Research Institute), Rome.

Glowka, L. (1998). A Guide to designing legal frameworks to determine access to genetic resources. Environmental Policy and Law Paper No. 34. IUCN, Gland.

Glowka, L. (2001). Towards a certification system for bioprospecting activities: Study commissioned by the State Secretariat for Economic Affairs. Sekretariat Leistungsbereich Wirtschaftspolitische Grundlagen, Staatssekretariat für Wirtschaft (SECO), Bern.

Glowka, L., T. Plän, and P.-T. Stoll (1997). Best practices for access to genetic resources. Background paper for the Correspondent Workshop on "Best Practices for Access to Genetic Resources" (co-sponsored by DG XI, European Commission and the German Federal Ministry of the Environment, Nature Conservation and Nuclear Safety), 14–17 January 1998, Cordoba.

Goeschl, T., and T. Lin (2004). Biodiversity conservation on private lands: Information problems and regulatory choices. Nota di lavoro 2004.55. Fondazione Eni Enrico Mattei, Milan.

Goeschl, T., and T. Swanson (1996). Market integration and crop genetic resource conservation. Paper prepared at the request of the International Plant Genetic Resources Institute (IPGRI), Faculty of Economics, Cambridge University, September 1996.

Goeschl, T., and T. Swanson (2002a). The social value of biodiversity for R&D. *Environmental and Resource Economics* 22(4): 477–504.

Goeschl, T., and T. Swanson (2002b). Lost horizons: The noncooperative management of an evolutionary biological system. Nota di lavoro 89.2002. Fondazione Eni Enrico Mattei, Milan.

Goeschl, T., and T. Swanson (2003a). Pests, plagues, and patents. *Journal of the European Economic Association* 1(2–3): 561–575.

Goeschl, T., and T. Swanson (2003b). On biology and technology: The economics of managing biotechnologies. Nota di lavoro 42.2003. Fondazione Eni Enrico Mattei, Milan.

Gollin, D., and R. Evenson (2003). Valuing animal genetic resources: Lessons from plant genetic resources. *Ecological Economics* 45(3): 353–363.

Gollin, D., M. Smale, and B. Skovmand (2000). Searching an ex situ collection of wheat genetic resources. *American Journal of Agricultural Economics* 82(4): 812–827.

Gonzâlez-Montagut, R. (2003). Private-public collaboration in funding protected areas in Mexico. Paper presented at the IUCN 5th World Parks Congress on Institutional Arrangements for Financing Protected Areas, 8–17 September, Durban.

Görg, C., and U. Brand (2000). Global environmental politics and competition between nation-states: On the regulation of biological diversity. *Review of International Political Economy* 7(3): 371–398.

Gowdy, J.M. (1997). The value of biodiversity: Markets, society and ecosystems. *Land Economics* 73(2): 25–41.

Green, M.J.B., and J. Paine (1997). State of the World's protected areas at the end of the twentieth century. Paper presented at the IUCN World Commission on Protected Areas Symposium: Protected Areas in the 21st Century : From Islands to Networks, 24–29 November, Albany, Australia.

Gren, I.-M., C. Folke, K. Turner, and I. Bateman (1994). Primary and secondary values of wetland ecosystems. *Environmental and Resource Economics* 4(1): 55–74.

Grifo, F., D.J. Newman, A.S. Fairfield, B. Bhattacharya, and J.T. Grupenhoff (1997). The origin of prescription drugs. In F. Grifo and J. Rosenthal (eds.), *Biodiversity and human health.* Washington, D.C: Island Press.

Grossman, S., and O. Hart (1986). The costs and benefits of ownership: A theory of vertical and lateral integration. *Journal of Political Economy* 94(4): 691–719.

Guerin-McManus, M., L.M. Famolare, I.A. Bowles, S.A.J. Malone, R.A. Mittermeier, and A.B. Rosenfeld (1998). Bioprospecting in practice: A case study of the Suriname ICBG Project and benefits sharing under the Convention on Biological Diversity. Available at <http://www.biodiv.org/doc/case-studies/abs/cs-abs-sr.pdf>, accessed on December 30, 2005.

Hampicke, U. (1999). The limits to economic valuation of biodiversity. *International Journal of Social Economics* 26(1/3): 158–173.

Hanley, N., C. Spash, and L. Walker (1995). Problems in valuing benefits of biodiversity protection. *Environmental and Resource Economics* 5(3): 249–272.

Hardin, G. (1968). The tragedy of the commons. *Science* 162: 1243–1248.

Harrison, J. (2002). International agreements and programmes on protected areas. *Parks* 12(3): 1–6.

Hart, O., and J. Moore (1990). Property rights and the nature of the firm. *Journal of Political Economy* 98(6): 1119–1158.

Hartwick, J.M. (1992). Deforestation and national accounting. *Environmental and Resource Economics* 2(5): 513–521.

Hawtin, G. (2004). The Global Crop Diversity Trust: Purposes, priorities and governance. In Swiss Agency for Development and Cooperation and the Syngenta Foundation for Sustainable Agriculture (eds.), *World Food Day 2004. Biodiversity for food security*. Basel.

Heal, G. (1995). Markets and biodiversity. Paine Webber Working Paper Series in Money, Economics and Finance PW 95-17. Columbia Business School, New York.

Heal, G. (1999b). New strategies for the provision of global public goods. Learning from international environmental challenges. In I. Kaul, I. Grunberg, and M.A. Stern (eds.), *Global public goods: International cooperation in the 21st century*. New York: Oxford University Press.

Heal, G. (1998). *Valuing the future: Economic theory and sustainability*. New York: Columbia University Press.

Heal, G. (2000). Valuing ecosystem services. *Ecosystems* 3(1): 24–30.

Heal, G. (2002). Globalization and biodiversity. Nota di lavoro 2002.101. Fondazione Eni Enrico Mattei, Milan.

Heal, G. (2003). Bundling biodiversity. *Journal of the European Economic Association* 1(2/3): 553–560.

Heal, G. (2004). Economics of biodiversity: An introduction. *Resource and Energy Economics* 26(2): 105–114.

Heal, G. and A.A. Small (2002). Agriculture and ecosystem services. In B.L. Gardner and G.C. Rausser (eds.), *Handbook of agricultural economics*. Vol. 2. Amsterdam: Elsevier.

Heal, G., B. Walker, S. Levin, K. Arrow, P. Dasgupta, G. Daily, P. Ehrlich, K.-G. Mäler, N. Kautsky, J. Lubchenco, S. Schneider, and D. Starrett (2003). Genetic diversity and interdependent crop choices in agriculture. *Resource and Energy Economics* 26(2): 175–184.

Helfer, L.R. (2005). Using intellectual property rights to preserve the global genetic commons: The International Treaty on Plant Genetic Resources for Food and Agriculture. In J.H. Riechman and K. Maskus (eds.), *International public goods and transfer of technology under a globalized intellectual property regime*. Cambridge. Forthcoming.

Heltberg, R. (2002). Property rights and natural resource management in developing countries. *Journal of Economic Surveys* 16(2): 189–214.

Henry, C. (1974). Option values in economics of irreplaceable assets. *Review of Economics Studies* 41: 89–104.

Hill, A.B. (1999). Trends in the international market for genetic and biochemical resources: Opportunities for Colombia. A report presented to the BIOTRADE Colombia Initiative Workshop, March 23–25, Villa de Leyva, Colombia. Washington, D.C.: WFED (World Foundation for Environment and Development).

Holm-Müller, K. (1999). Finanzierung von Naturschutz über private Komplementärgüter. *Konjunkturpolitik (Applied Economics Quarterly)* 45(1): 24–39.

Hotte, L., N. Van Long, and H. Tian (2000). International trade with endogenous enforcement of property rights. *Journal of Development Economics* 62(1): 25–54.

Huybers, T., and J. Bennet (2003). Environmental management and the competitiveness of nature-based tourism destinations. *Environmental and Resource Economics* 24(3): 213–233.

ICUN (The World Conservation Union) (1994): Guidelines for protected area management categories. CNPPA with assistance of WCMC. Gland, Switzerland, and Cambridge.

Ihori, T. (1996). International public goods and contribution productivity differentials. *Journal of Public Economics* 61(1): 139–154.

Immordino, G. (2003). Looking for a guide to protect the environment: The development of the precautionary principle. *Journal of Economic Surveys* 17(5): 629–643.

IMS Health (2005). 2002 World pharma sales growth: Slower, but still healthy. Available at <http://www.imshealth.com/web/end/0,3150,64576068_63872702_70514186,00.html>, accessed on December 30, 2005.

Innes, R. (2000). The economics of takings and compensation when land and its public use value are in private hands. *Land economics* 76(2): 195–212.

Innes, R., S. Polasky, and J. Tschirhart (1998). Takings, compensation and endangered species protection on private lands. *The Journal of Economic Perspectives* 12(3): 35–52.

IPGRI (International Plant Genetic Resources Institute) (1996). Access to plant genetic resources and the equitable sharing of benefits: A contribution to the debate on systems for the exchange of germplasm. Issues in Genetic Resources No. 4, June 1996. Rome: International Plant Genetic Resources Institute.

IPGRI (International Plant Genetic Resources Institute) (1999). Key questions for decisionmakers. Protection of plant varieties under the WTO Agreement on Trade-Related Aspects of Intellectual Property Rights. Decision Tools October 1999. Rome: International Plant Genetic Resources Institute.

Irwin, E.G., and N.E. Bockstael (2002). Interacting agents, spatial externalities and the evolution of residential land use patterns. *Journal of Economic Geography* 2: 31–54.

ISF (International Seed Federation) (2005). Seed statistics. World seed trade statistics. Available at <http://www.worldseed.org/statistics.htm>, accessed on December 30, 2005.

Itaya, J.-I., and N. Yamada (2004). Private Provision of Public Goods in Renegotiation-Proof Equilibria. Discussion Paper Series A 95. Hokkaido University.

James, A.N., K.J. Gaston, and A. Balmford (1999b). Balancing the Earth's accounts. *Nature* 401: 323–324.

James, A.N., M.J.B. Green, and J.R. Paine (1999a). A global review of protected area budgets and staffing. WCMC Biodiversity Series No. 10. World Conservation Monitoring Centre, Cambridge.

Janssen, J. (1999). Property rights on genetic resources: Economic issues. *Global Environmental Change* 9(4): 313–321.

Jayaraman, R., and R. Kanbur (1999). International public goods and the case for foreign aid. In I. Kaul et al. (eds.), *Global public goods: International cooperation in the 21st century*. New York: Oxford University Press.

Jürgens, N. (2001). Biodiversity: The living resource: Challenges and research strategies. Bonn: NKGCF (Nationales Kommitee für Global Change Forschung/German National Committee on Global Change Research).

Kanbur, R. (2002). International financial institutions and international public goods: Operational implications for the World Bank. G-24 Discussion Paper Series No. 19. United Nations Conference on Trade and Development, United Nations Center for International Development, Harvard University.

Kaplow, L., and S. Shavell (1999): Economic analysis of law. NBER Working Paper No. 6960. National Bureau of Economic Research, Cambridge, Mass.

Karp, L., S. Sacheti, and J. Zhao, (2001). Common ground between free-traders and environmentalists. *International Economic Review* 42(3): 617–648.

Kassar, I., and P. Lasserre (2004). Species preservation and biodiversity value: A real options approach. *Journal of Environmental Economics and Management* 48(2): 857–879.

Kaufmann, D., A. Kraay, and P. Zoido-Lobatón (1999). Aggregating governance indicators. Policy Research Working Paper 2195. World Bank Development Research Group Macroeconomics and Growth and World Bank Institute Governance, Regulation and Finance, Washington, D.C.

Kaul, I., and R.U. Mendoza (2003). Advancing the concept of public goods. In I. Kaul, P. Coneicao, K. Le Goulven, and R.U. Mendoza (eds.), *Providing global public goods: Managing Globalization*. New York: United Nations Development Programme.

Kaul, I., I. Grunberg, and M.A. Stern (1999). Defining global public goods. In I. Kaul et al. (eds.), *Global public goods—International cooperation in the 21st century*. New York: Oxford University Press.

Kemp, M.C. (1984). A note of the theory of international transfers. *Economics Letters* 14(2–3): 259–262.

Khanna, J. (1993). Cooperative versus noncooperative behavior: The case of agricultural research. *Review of Economics and Statistics* 75(2): 346–352.

Kiesling, H.J. (1974). Public goods and possibilities for trade. *Canadian Journal of Economics* 7(3): 402–417.

Klemperer, P. (1990). How broad should the scope of patent protection be? Discussion paper series No. 392. Centre for Economic Policy Research, London.

Kloppenburg, J.R., Jr., and D.L. Kleinman (1988). Seeds of controversy: National property versus common heritage. In J.R. Kloppenburg Jr. (ed.), *Seeds and sovereignty. The use and control of plant genetic resources.* Durham: Duke University Press.

Kölle, C. (1995): *Ökonomische Analyse internationaler Umweltkooperationen.* Heidelberg: Physica-Verlag.

Kolstad, C.D. (2000). *Environmental Economics.* New York: Oxford University Press.

Koo, B., and B.D. Wright (2002). The timing of evaluation of genebank materials and the effects of biotechnology. In C. Moss et al. (eds.), *Agricultural globalization, trade, and the environment.* Boston, Mass.: Kluwer Academic Publications.

Koo, B., P.G. Pardey, and B.D. Wright (2003). The economic costs of conserving genetic resources at the CGIAR centres. *Agricultural Economics* 29(3): 287–297.

Koopman, J. (2005). Reconciliation of proprietary interests in genetic and knowledge resources: Hurry cautiously! *Ecological Economics* 54(4): 523–541.

Kotchen, M.J. (2005). Impure public goods and the comparative statics of environmentally friendly consumption. *Journal of Environmental Economics and Management* 49(2): 281–300.

Krautkraemer, J.A. (1985). Optimal growth, resource amenities and the preservation of natural environments. *Review of Economic Studies* 52(1): 153–170.

Kremer, M., and C. Morcom (2000). Elephants. *The American Economic Review* 90(1): 212–234.

Krutilla, J.V. (1967). Conservation reconsidered. *The American Economic Review* 57(4): 777–786.

Kulessa, M.E. and M. Ringel (2003). Kompensationen als innovatives Instrument globaler Umweltschutzpolitik. Möglichkeiten und Grenzen einer Weiterentwicklung am Beispiel der biologischen Vielfalt. *Zeitschrift für Umweltpolitik und Umweltrecht* 26(3): 263–285.

Kumari, K., and K. King (1997). Paradigm cases to illustrate the application of the incremental cost assessment to biodiversity. PRINCE. GEF Secretariat. Washington, D.C.

Kwon, G.-H. (1998). Retests on the theory of collective action and the Olson and Zeckhauser model and its elaboration. *Economics and Politics* 10(1): 37–62.

Laffont, J.-J. (1988). *Fundamentals of public economics.* Cambridge, Mass.: MIT Press.

Laird, S.A. (1993). Contracts for Biodiversity Prospecting. In W.V. Reid, S.A. Laird, C.A. Meyer, R. Gamez, A. Sittenfeld, D.H. Janzen, D.H. Gollin, and C. Juma (eds.), *Biodiversity Prospecting: Using Genetic Resources for Sustainable Development.* Washington, D.C.: World Resource Institute.

Laird, S.A. (2000). Conservation Challenges in the Botanical Medicine Industry. *Rainforest Medical Bulletin* 7(1).

Laird, S.A., and R. Wynberg (2003). Biodiversity prospecting and access and benefit-sharing: An introductory primer. Dept. for International Development United Kingdom, IUCN South Africa. Deutsche Gesellschaft für Technische Zusammenarbeit. Gland: IUCN.

Laird, S.A., S. Johnson, R. Wynberg, E. Lisinge, and D. Lohan (2003). Biodiversity access and benefit-sharing policies for protected areas: An introduction. UNU/IAS Report. Tokyo.

Lambert, A. (2003). Introduction to the establishment of environmental trust funds. Paper prepared for the International Academy for the Environment (BfN) workshop on "Introduction to Environmental Funds in Kazakhstan, Kyrgyzstan and Georgia," 13–17 April 2003, Vilm, Germany.

Lange, D. (1996). Untersuchungen zum Heilpflanzenhandel in Deutschland. Ein Beitrag zum internationalen Artenschutz. Hiltrup.

Lankoski, J., and M. Ollikainen (2003). Agri-environmental externalities: A framework for designing targeted policies. *European Review of Agricultural Economics* 30(1): 51–75.

LaPage, W.F. (1994). America's self-financing park system: The New Hampshire Model. In M. Munashinghe and J. McNeely (eds.), *Protected area economics and policy—Linking conservation and sustainable development.* Washington, D.C.: IUCN.

Lapham, N.P., and R.J. Livermore (2003): Striking a balance. Ensuring conservation's place on the international biodiversity assistance agenda. Center for Applied Biodiversity Science, Center for Conservation and Government, Conservation International, Washington, D.C.

Laurance, W.F. (2004). The perils of payoff: Corruption as a threat to global biodiversity. *Trends in Ecology and Evolution* 19(8): 399–401.

Laxminarayan, R., and G. Brown (2001). Economics of antibiotic resistance: A theory of optimal use. *Journal of Environmental Economics and Management* 42(2): 183–206.

Laxminarayan, R., and R.D. Simpson (2002). Refuge strategies for managing pest resistance in transgenic agriculture. *Environmental and Resource Economics* 22(4): 521–536.

Le Breton, G. (2001). Trade in biological resources in southern Africa. Paper presented to the Multi-Stakeholder Dialogue on Trade, Intellectual Property Rights and Biological Resources in Eastern and Southern Africa, 31 July 2001, Nyeri. Geneva: ICTSD (The International Centre for Trade and Sustainable Development).

Lerch, A. (1994). Property rights und biologische Vielfalt. *Zeitschrift für Umweltpolitik und Umweltrecht* 17(3): 289–303.

Lerch, A. (1998). Property rights and biodiversity. *European Journal of Law and Economics* 6(3): 285–304.

Lerch, A. (2000). Intellectual property rights and genetic resources: An economic analysis. Volkswirtschaftliche Diskussionsbeiträge 08/00. Fachbereich Wirtschaftswissenschaften, Gesamthochschule Kassel.

Levin, S.A., S. Barrett, S. Aniyar, W. Baumol, C. Bliss, B. Bolin, P. Dasgupta, P. Ehrlich, C. Folke, I.-M. Gren, C.S. Holling, A. Jansson, B.-O. Jansson, K.-G. Mäler, D. Martin, C. Perrings, and E. Shesinski (1998). Resilience in natural and socio-economic systems. *Environment and Development Economics* 3(2): 222–235.

Lewandrowski, J., R.F. Darwin, M. Tsigas, and A. Raneses (1999). Estimating costs of protecting global ecosystem diversity. *Ecological Economics* 29(1): 111–125.

Li, C.-Z., and K.-G. Löfgren (1998). A dynamic model of biodiversity preservation. *Environment and Development* 3(2): 157–172.

Liebig, K., D. Alker, K. ould Chih, H. Illi, and J. Wolf (2003). Governing biodiversity: Access to genetic resources and approaches to obtaining benefits from their use: The case of the Philippines. Bonn: German Development Institute.

Linarelli, J. (2004). Treaty governance, intellectual property and biodiversity. *Environmental Law Review* 6(1): 21–38.

Lindberg, K. (2003). The "sale" of biodiversity to nature tourists. Working Party on Global and Structural Policies/Working Group on Economic Aspects of Biodiversity. *ENV/EPOC/GSP/BIO(2001)10/FINAL*. Unclassified. OECD (Organisation for Economic Co-operation and Development), Paris.

Linden, E., T. Lovejoy, and J.D. Phillips (2004). Seeing the forest: Conservation on a continental scale. *Foreign Affairs* 83(4): 8–13.

Lomborg, B. (2001). *The skeptical environmentalist: Measuring the real state of the world.* Cambridge: Cambridge University Press.

Maier, L., and M. Shobayashi (2001). Multifunctionality: Towards an analytical framework. Paris: OECD.

Maignan, C., G.I.P. Ottaviano, D. Pinelli, and F. Rullani (2003). Bio-ecological diversity vs. socio-economic diversity: A comparison of existing measures. Nota di lavoro 2003.13. Fondazione Eni Enrico Mattei, Milan.

Mainwaring, L. (2001). Biodiversity, bio-complexity and the economics of genetic dissimilarity. *Land Economics* 77(1): 79–83.

Mäler, K.-G. (2000). Development, ecological resources and their management: A study of complex dynamic systems. *European Economic Review* 44(4/6): 645–665.

Marcoux, A. (2000). Population and deforestation. FAO. Population and the environment: A review and concepts for population programmes. Part III: Population and deforestation. *SD Dimensions* Sustainable Development Department, Food and Agriculture Organization of the United Nations. June 2000. Rome.

Margules, C.R., and R.L. Pressey (2000). Systematic conservation planning. *Science* 405: 243–253.

Mas-Colell, A., M.D. Whinston, and J.R. Green (1995): *Microeconomic theory.* New York: Oxford University Press.

Maskus, K.E. (2000). Intellectual property rights and foreign direct investment. Policy Discussion Paper No. 0022. Centre for International Economic Studies, University of Adelaide.

Maskus, K.E. (2005). The role of intellectual property rights in encouraging foreign direct investment. In C. Fink (ed.), *Intellectual property and development: Lessons from recent economic research*. Washington, D.C.

Mateo, N., W. Nader, and G. Tamayo (2001). Bioprospecting. In S.A. Levin and R.E. Colwell (eds.), *Encyclopedia of biodiversity*. Volume 1. San Diego: Academic Press.

Matz, N. (2003). Protected areas in international nature conservation law: Can states obtain compensation for their establishment? *Zeitschrift für ausländisches öffentliches Recht und Völkerrecht* 63(3).

McAusland, C., and C. Costello (2004). Avoiding invasives: Trade-related policies for controlling unintentional exotic species introductions. *Journal of Environmental Economics and Management* 48(2): 954–977.

McCann, K. (2000). The diversity-stability debate. *Science* 405: 228–233.

McCann, L., B. Colby, K.W. Easter, A. Kasterine, and K.V. Kuperan (2005). Transaction cost measurement for evaluating environmental policies. *Ecological Economics* 52(4): 527–542.

McChesney, J. (1996). Biological diversity, chemical diversity and the search for new pharmaceuticals. In J.B. Balick, E. Elisabetsky, and S.A. Laird (eds.), *Medicinal resources of the tropical forest: Biodiversity and its importance to human health*. New York: Columbia University Press.

McGuire, M.C., and C.H. Groth (1985). A method for identifying the public good allocation. *Quarterly Journal of Economics* 100(Supplement): 915–934.

McNeely, J.A. (1997). Sustainable finance for protected areas: The World Conservation Union. Paper presented at "Protected Areas in the 21st Century: From Islands to Networks" workshop, 24–28 November, Albany, Australia.

McNeely, J.A., J. Harrison, and P. Dingwall (eds.) (1994). *Protecting nature: Regional views of protected areas*. Gland: IUCN.

McNeely, J.A., K.R. Miller, W.V. Reid, R.A. Mittermeier, and T.B. Werner (1990). Conserving the World's Biological Diversity. Gland: World Conservation Union and Washington, D.C.: World Resources Institute, Conservation International, World Wide Fund for Nature, and the World Bank.

MEA (Millennium Ecosystem Assessment) (2005a). Ecosystems and human well-being: General Synthesis. Washington, D.C.: World Resources Institute

MEA (Millennium Ecosystem Assessment) (2005b). Ecosystems and human well-being: Biodiversity Synthesis. Washington, D.C.: World Resources Institute.

Mendelsohn, R. (1994). Property rights and tropical deforestation. *Oxford Economic Papers* 46(Supplement): 750–756.

Mendelsohn, R. (2003). The challenge of conserving indigenous domesticated animals. *Ecological Economics* 45(3): 501–510.

Mendelsohn, R., and M.J. Balick (1995). The value of undiscovered pharmaceuticals in tropical forests. *Economic Botany* 49: 223–228.

Mensink, P., and T. Requate (2005). The Dixit-Pindyck and the Arrow-Fisher-Hanemann-Henry option values are not equivalent: A note on Fisher (2000) *Resource and Energy Economics* 27(1): 83–88.

Metrick, A., and M. Weitzman (1998). Conflicts and choices in biodiversity preservation. *Journal of Economic Perspectives* 12(3): 21–34.

Mikitin, K. (1995). Issues and options in the design of GEF supported trust funds for biodiversity conservation. Environment Department Papers, Biodiversity Series. World Bank, Washington, D.C.

Miller, K.N. (1984): The Bali Action Plan: A framework for the future of protected areas. In J.A. McNeely and K.R. Miller (eds.), *National parks, conservation and development: The role of protected areas in sustaining society*. Washington, D.C.: Smithsonian Institute Press.

Mohr, E. (1990). Burn the forest!: A bargaining theoretic analysis of a seemingly perverse proposal to protect the rainforest. Kiel Working Paper No. 447. Kiel Institute for World Economics, Kiel.

Moran, D., and D. Pearce (1997). The economics of biodiversity. In H. Folmer and T. Tietenberg (eds.), *The international yearbook of environmental and resource economics 1997/1998: A survey of current issues*. Cheltenham: Edward Elgar.

Moran, D., D. Pearce, and A. Wendelaar (1996). Global biodiversity priorities: A cost-effectiveness index for investments. *Global Environmental Change* 6(2): 103–119.

Mugabe, J. (1998). Intellectual property protection and traditional knowledge: An exploration in international policy discourse. Paper prepared by African Centre for Technology Studies for the World Intellectual Property Organization.

Mulholland, D.M., and E.A. Wilman (2003). Bioprospecting and biodiversity contracts. *Environment and Development Economics* 8(3): 417–435.

Muller, J., and H.J. Albers (2004). Enforcement, payments, and development projects near protected areas: How the market setting determines what works where. *Resource and Energy Economics* 26(2): 185–204.

Mulongoy, K.J., and S. Chape (2002). Protected areas and biodiversity: An overview of key issues. Cambridge: UNEP World Conservation Monitoring Centre.

Munro, A. (1997). Economics and biological evolution. *Environmental and Resource Economics* 9(4): 429–449.

Murdoch, J.C., and T. Sandler (1997). The voluntary provision of a pure public good: The case of reduced CFC emission and the Montreal Protocol. *Journal of Public Economics* 63(2): 331–349.

Murtough, G., B. Aretino,, and A. Matysek (2002). Creating markets for eco-system services. Productivity Commission Staff Research Paper. AusInfo, Canberra.

Musgrave, R.A. (1993). Tax equity with multiple jurisdictions. In Musgrave, R.A. (2004). *Public finance in a democratic society*. Volume II. Cheltenham.

Myers, N., R.A. Mittermeier, C.G. Mittermeier, G.A.B. da Fonseca, and J. Kent (2000). Biodiversity hotspots for conservation priorities. *Nature* 403: 853–858.

Naidoo, R., and W.L. Adamowicz (2005). Biodiversity and nature-based tourism at forest reserves in Uganda. *Environment and Development Economics* 10(2): 159–178.

Nehring, K., and C. Puppe (2002). A theory of diversity. *Econometrica* 70(3): 1155–1198.

Nehring, K., and C. Puppe (2004). Modelling phylogenetic diversity. *Resource and Energy Economics* 26(2): 205–235.

Newell, R.G., J.N. Sanchirico, and S. Kerr (2005). Fishing quota markets. *Journal of Environmental Economics and Management* 49(3): 437–462

North, D.C. (1990). *Institutions, institutional change and economic performance.* Cambridge: Cambridge University Press.

Nottenburg, C., P.G. Pardey, and B.D. Wright (2003). Accessing other people's technology. Brief 4:1-7. Research at a glance: Biotechnology and Genetic Resource Policies. IFPRI (International Food Policy Research Institute), Rome.

Nunes, P.A.L.D., and Y.E. Riyanto (2001). Policy instruments for creating markets for biodiversity: Certification and ecolabing. Nota di lavoro 2001.72. Fondazione Eni Enrico Mattei, Milan.

Nunes, P.A.L.D., and J.C.J.M van den Bergh (2001). Economic valuation of biodiversity: Sense or nonsense? *Ecological Economics* 39(2): 203–222.

Nunes, P.A.L.D., J.C.J.M van den Bergh, and P. Nijkamp (2001). Ecological-economic analysis and valuation of biodiversity. Nota di lavoro 2001.74. Fondazione Eni Enrico Mattei, Milan.

Nunnenkamp, P. (1992). International financing of environmental protection. North-South conflicts on concepts and financial instruments and possible solutions. Kiel Working Paper No. 512. Kiel Institute for World Economics, Kiel.

OECD (Organisation for Economic Co-operation and Development) (1996). Intellectual property, technology transfer and genetic resources: An OECD survey and current practices and polices. Paris.

OECD (Organisation for Economic Co-operation and Development) (1997). Issues in the sharing of benefits arising out of the utilisation of genetic resources. Unclassified Paper OCDE/GP(97)193. Paris.

OECD (Organisation for Economic Co-operation and Development) (1998a). Biotechnology for clean industrial products and processes: Towards industrial sustainability. Paris.

OECD (Organisation for Economic Co-operation and Development) (1998b). Economic issues in Benefit Sharing: Concepts and Practical Experiences [ENV/EPOC/GEEI/BIO(98)7/FINAL]. Paris.

OECD (Organisation for Economic Co-operation and Development) (1999a). Handbook of incentive measures for biodiversity. Design and implementation. Paris.

OECD (Organisation for Economic Co-operation and Development) (1999b). Economic issues in benefit sharing: Concepts and practical experiences. Working party on Economic and Environmental Policy Integration. Working group on Economic Aspects of Biodiversity. ENV/EPOC/GEEI/BIO(98)7/Final. Paris.

OECD (Organisation for Economic Co-operation and Development) (2002). Genetic inventions, intellectual property rights and licensing practices: Evidence and policies. Paris.

OECD (Organisation for Economic Co-operation and Development) (2003a). Harnessing markets for biodiversity. Towards conservation and sustainable use. Paris.

OECD (Organisation for Economic Co-operation and Development) (2003b). Economic issues in access and benefit sharing of genetic resources: A framework of the analysis. Working party on global and structural policies. Working group on Economic Aspects of Biodiversity. ENV/EPOC/GSP/BIO(2001)2/Final. Paris.

OECD (Organisation of Economic Cooperation and Development) (2003c). Philanthropic foundations and development Co-operation. Off-print of the DAC Journal 2003, 4(3). Paris.

OECD (Organisation for Economic Co-operation and Development) (2004). Handbook of market creation for biodiversity: Issues in implementation. Paris.

OECD (Organisation for Economic Co-operation and Development) (2005). Statistics> National Accounts>Input-output Tables. Available at <http://www.oecd.org/document/6/0,2340,en_2825_495684_2672966_1_1_1_1,00.html>, accessed on December 30, 2005.

Oldfield, M.L. (1984). The value of conserving genetic resources. U.S. Department of the Interior, National Park Service, Washington, D.C.

Olson, M. Jr. (1965). *The logic of collective action: Public goods and the theory of groups.* Cambridge, Mass.: Harvard University Press.

Olson, M. Jr., and R. Zeckhauser (1966). An economic theory of alliances. *Review of Economics and Statistics* 48(3): 266–279.

OTA (Office of Technology Assessment) (1991). Biotechnology in a global economy. OTA-BA-494. U.S. Congress, Washington, D.C.

Øygard, R., and D.W. Bromley (1998). The Global Environmental Facility: Managing side payments in global environmental accords. Working paper. Noragric, As. University of Wisconsin, Madison.

Paarlberg, P.L., M. Bredahl, and J.G. Lee (2002). Multifunctionality and agricultural trade negotiations. *Review of Agricultural Economics* 24(2): 322–335.

Pachico, D. (2001). Implementing farmers' rights in genetic resources: Approaches in benefit sharing. *Biopolicy* 4(1).

Pagiola, S., K. von Ritter, and J. Bishop (2004). Assessing the economic value of ecosystem conservation. *Environment Department Paper No. 101.* World Bank Environment Department, in collaboration with The Nature Conservancy and IUCN-World Conservation Union, Washington, D.C.

Painting, K.A., M.C. Perry, R.A. Denning, and W.G. Ayad (1993). *Guidebook for genetic resources documentation.* Rome: IPGRI (International Plant Genetic Resources Institute).

Panayotou, T. (1994). Conservation of biodiversity and economic development: The concept of transferable development rights. *Environmental and Resource Economics* 4(1): 95–110.

Pardey, P.G., B.D. Wright, C. Nottenburg, E. Binenbaum, and P. Zambrano (2003). Intellectual property and developing countries: Freedom to operate in agricultural biotechnology. Brief 3:1-6. Research at a glance: Biotechnology and Genetic Resource Policies. IFPRI (International Food Policy Research Institute), Rome.

Park, P.J., E.B. Barbier, and J.C. Burgess (1998). The economics of forest land use in temperate and tropical areas. *Environmental and Resource Economics* 11(3–4): 473–487.

Pearce, D. (1995). The Global Environment Facility and the Rio Conventions. In D. Pearce (ed.), *Capturing global environmental value*. London: Earthscan.

Pearce, D. (1998). *Economics and environment: Essays on ecological economics and sustainable development.* Cheltenham: Edward Elgar.

Pearce, D., and C. Perrings (1995). Biodiversity conservation and economic development: Local and global dimensions. In C. Perrings et al. (eds.), *Biodiversity conservation: Problems and policies.* Papers from the Biodiversity Programme. Beijer International Institute of Ecological Economics, Royal Swedish Academy of Sciences. Stockholm.

Pearce, D., and S. Puroshothaman (1995). The economic value of plant-based pharmaceuticals. In T. Swanson (ed.), *Intellectual property rights and biodiversity conservation: An interdisciplinary analysis of the values of medicinal plants.* Cambridge: Cambridge University Press.

Pearce, D.G. (1992). Repeated games: Cooperation and rationality. In J.-J. Laffont (ed.), *Advances in Economic Theory: Sixth World Congress. Vol. I.* Cambridge: Cambridge University Press.

Perrings, C. (1998). Resilience in the dynamics of economy-environment systems. *Environmental and Resource Economics* 11(3–4): 503–520.

Perrings, C., and M. Gadgil (2003). Conserving biodiversity: Reconciling local and global benefits. In I. Kaul, P. Coneicao, K. Le Goulven, and R.U. Mendoza (eds.), *Providing global public goods: Managing Globalization.* New York: United Nations Development Programme.

Perrings, C., and H. Opschoor (1994). The loss of biological diversity: Some policy implications. *Environmental and Resource Economics* 4(1): 1–11.

Perrings, C., and D. Pearce (1994). Threshold effects and incentives for the conservation of biodiversity. *Environmental and Resource Economics* 4(1): 13–28.

Perrings, C., and B. Walker (2004). Conservation in the optimal use of rangelands. *Ecological Economics* 49(2): 119–128.

Perrings, C., K.-G. Mäler, C. Folke, C.S. Holling, and B.-O. Jansson (1995). Biodiversity conservation and economic development: The policy problem. In C. Perrings et al. (eds.), *Biodiversity conservation: Problems and policies.* Papers from the Biodiversity Programme, Beijer International Institute of Ecological Economics, Royal Swedish Academy of Sciences, Stockholm.

Peterson, J.M., R.N Boisvert, and H. de Gorter (2002). Environmental policies for a multifunctional agricultural sector in open economies. *European Review of Agricultural Economics* 29(4): 423–443.

PhRMA (Pharmaceutical Research and Manufacturers of America) (2003). *Pharmaceutical Industry Profile 2003.* Washington, D.C.

Plotkin, J.B., M.D. Potts, D.W. Yu, S. Bunyavejchewin, R. Condit, R. Foster, S. Hubbell, J. LaFrankie, N. Manokaran, H.-S. Lee, R. Sukumar, M.A. Nowak, and P.S. Ashton (2000). Predicting species diversity in tropical forests. *PNAS* 97(20): 10850–10854.

Polasky, S., and H. Doremus (1998). When the truth hurts: Endangered species policy on private land with imperfect information. *Journal of Environmental Economics and Management* 35(1): 22–47.

Polasky, S., and A. Solow (1995). On the value of a collection of species. *Journal of Environmental Economics and Management* 29(3): 298–303.

Polasky, S., J.D. Camm, and B. Garber-Yonts (2001). Selecting biological reserves cost-effectively: An application to terrestrial vertebrate conservation in Oregon. *Land Economics* 77(1): 68–78.

Polasky, S., C. Costello, and C. McAusland (2005). On trade, land-use, and biodiversity. *Journal of Environmental Economics and Management* 48(2): 911–925.

Prakash, S. (2000). WTO rules: Do they conserve or threaten biodiversity? *The Journal of World Intellectual Property* 3(1): 155–165.

Principe, P. (1989). *The economic value of biodiversity among medicinal plants.* Paris: OECD.

Purvis, H., and A. Hector (2000). Getting the measure of biodiversity. *Science* 405: 212–219.

Quintela, C.E., L. Thomas, and S. Robin (2004). Proceedings of the Workshop "Stream Building a Secure Financial Future: Finance & Resources." 5th IUCN World Parks Congress. IUCN, Gland, Switzerland and Cambridge.

Ramanna, A., and M. Smale (2004). Rights and access to plant genetic resources under India's new law. *Development Policy Review* 22(4): 423–442.

Ramsar (Ramsar Convention on Wetlands) (2002): Ramsar COP8 DOC. 17: Endowment fund for Ramsar SGF. Available at <http://www.ramsar.org/cop8/cop8_doc_17_e.htm>, accessed on December 30, 2005.

Rausser, G.C., and A.A. Small (1996). The economic value of patents, licenses, and plant variety protection. Working Paper No.797. Department of Agricultural and Resource Economics, University of California, Berkeley.

Rausser, G.C., and A.A. Small (1998). Bioprospecting with patent races. Paine Webber Working Paper Series in Money, Economics and Finance PW-98-07. Columbia Business School, New York.

Rausser, G.C., and A.A. Small (2000). Valuing research leads: Bioprospecting and the conservation of genetic resources. *Journal of Political Economy* 108(1): 1–33.

Rausser, G.C., and A.A. Small (2001). Genetic resource libraries: Bioprospecting and knowledge assets. Paper presented at NCEAS Workshop on Economics of Biodiversity, 2001, Santa Barbara.

Reid, W.R. (1992). How many species will there be? In T.C. Whitmore and J.A. Spayer (eds.), *Tropical deforestation and species extinction.* New York: Chapman and Hall (reproduced by CIESIN).

Reid, W.V., S.A. Laird, C.A. Meyer, R. Gamez, A. Sittenfeld, D.H. Janzen, D.H. Gollin, and C. Juma (1993). *Biodiversity prospecting: Using genetic resources for sustainable development.* Washington, D.C.: World Resource Institute.

Reuters (2004). Major drugs industry: Industry statistics. Available at <http://www.investor.reuters.com/Industry Center.aspx?industrypscode=MAJRRX&target=%2findustries%2findhighlights%2findustrycenter>, accessed on October 16, 2004.

Richerzhagen, C. (2003). Ökonomische Analyse nationaler Zugangs- und Vorteilsausgleichsregelungen genetischer Ressourcen am Beispiel der Philippinen. In H. Korn and U. Feit (eds.), *Treffpunkt Biologische Vielfalt III.* Bonn: Bundesamt für Naturschutz.

Richerzhagen, C., and K. Holm-Mueller (2005): The effectiveness of access and benefit sharing in Costa Rica: Implications for national and international regimes. *Ecological Economics* 54(4): 445–460.

Ritter, W., and B. Kosak (1996). Bedingungen für den Zugang zu und die Nutzung von genetischen Ressourcen für die Ernährung und Landwirtschaft als Bestandteile der biologischen Vielfalt. Schriften zu Genetischen Ressourcen Informationszentrum Biologische Vielfalt (IBV) der ZADI: Zugang zu Pflanzengenetischen Ressourcen für die Ernährung und Landwirtschaft: der Diskussionsprozess in Deutschland 3: 92–101.

Romstad, E., A. Vatn, P.K. Rorstad, and V. Soyland (2000). Multifunctional agriculture: Implications for policy design. Report 21. Department of Economics and Social Sciences, University of Norway, Aas.

Roosen, J., A. Fadlaoui, and M. Bertaglia (2003). Economic evaluation and biodiversity conservation of animal genetic resources. FE Working paper 0304. Department of Food Economics and Consumption Studies, University of Kiel.

Root, T.L., D.P. MacMynowski, M.D. Mastrandrea, and S.H. Schneider (2005). Human-modified temperatures induce species changes: Joint attribution. PNAS 102(21): 7465–7469.

Rosenthal, J. (1996). The International Cooperative Biodiversity Groups (ICBG) Program. Fogarty International Center, National Institutes of Health. Available at <http://www.biodiv.org/doc/case-studies/abs/cs-abs-icbg.pdf>, accessed on December 30, 2005.

Rowthorn, B., and G. Brown (1999). When a high discount rate encourages biodiversity. *International Economic Review* 40(2): 315–332.

SAG (Schweizer Arbeitsgruppe Gentechnologie) (1997). Es ist doch ein Kraut gewachsen: Naturstoffe, synthetische Arzneimittel und Gentech-Medikamente im Vergleich. Zürich.

Sala, O.E., F.S. Chapin, III, J.J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L.F. Huenneke, R.B. Jackson, A. Kinzig, R. Leemans, D.M. Lodge, H.A. Mooney, M. Oesterheld, N.L. Poff, M.T. Sykes, B.H. Walker, M. Walker, and D.H. Wall (2000). Global biodiversity scenarios for the year 2100. *Science* 287: 1770–1774.

Samprath, P.G. (2000). Biodiversity prospecting contracts: Institutional and organizational issues in drug discovery and research. In BfN (Bundesamt für Naturschutz/German Federal Agency of Nature Conservation) (ed.), *Treffpunkt Biologische Vielfalt.* Bonn.

Sanchirico, J.N., and J.E. Wilen (1999). Bioeconomics of spatial exploitation in a patchy environment. *Journal of Environmental Economics and Management* 37(2): 129–150.

Sandler, T. (1992). *Collective action: Theory and applications.* New York: Harvester Wheatsheaf.

Sandler, T. (1993). Tropical deforestation: Markets and markets failures. *Land Economics* 69(3): 225–233.

Sandler, T. (2002). Financing international public goods. In M. Ferroni and A. Mody (eds.), *International public goods: Incentives, measurement, and financing.* World Bank. Washington, D.C.

Sandler, T., and J.F. Forbes (1980). Burden sharing, strategy and the design of NATO. *Economic Inquiry* 18(3): 425–444.

Sandler, T., and K. Hartley (2001). Economics of alliances: The lessons for collective action. *Journal of Economic Literature* 39(3): 869–896.

Sandler, T., and J.C. Murdoch (1990). Nash-Cournot of Lindahl behavior? An empirical test for the NATO allies. *Quarterly Journal of Economics* 105(4): 875–894.

Sandler, T., and J.C. Murdoch (2000). On sharing NATO defence burdens in the 1990s and beyond. *Fiscal Studies* 21(3): 297–327.

Sanjayan, M.A., and M.E. Soulé, (1997). Moving beyond Brundtland: The conservation value of British Colombia's 12 percent protected area strategy. Greenpeace. Preliminary Report. Available at <http://archive.greenpeace.org/comms/97/forest/soule.html>, accessed on December 30, 2005.

SCBD (Secretariat of the Convention on Biological Diversity) (2001). The value of forest ecosystems. SCBD CBD Technical Series no. 4. Montreal.

SCBD (Secretariat of the Convention on Biological Diversity) (2004). Biodiversity issues for consideration in the planning, establishment and management of protected area sites and networks. Montreal.

SCBD (Secretariat of the Convention on Biological Diversity) (2005). Programmes: Access to Genetic Resources and Benefit-sharing. Database on ABS Measures. Available at <http://www.biodiv.org/programmes/socio-eco/benefit/measures.aspx>, accessed on February 1, 2005.

Scheldon, J.W., and M.J. Balick (1995). Ethnobotany and the search for balance between use and conservation. In T. Swanson (ed.), *Intellectual property rights and biodiversity conservation: An interdisciplinary analyses of the values of medicinal plants.* Cambridge: Cambridge University Press.

Schlager, E., and E. Ostrom (1992). Property-rights regimes and natural resources: A conceptual analysis. *Land Economics* 68(3): 249–62.

Sedjo, R.A. (1988). Property rights and the protection of plant genetic resources. In J.R. Kloppenburg Jr. (ed.), *Seeds and sovereignty. The use and control of plant genetic resources.* Durham: Duke University Press.

Sedjo, R.A. (1992). Property rights, genetic resources, and biotechnological change. *Journal of Law and Economics* 35(2): 199–213.

Sedjo, R.A., and R.D. Simpson (1995). Property rights, externalities and biodiversity. In T. Swanson (ed.), *The economics and ecology of diversity decline: The forces driving global change.* Cambridge: Cambridge University Press.

Shafik, N. (1994). Economic development and environmental quality: An econometric analysis. *Oxford Economic Papers* 46(Supplement): 757–773.

Siebert, H. (1977). Environmental quality and the gains from trade. *Kyklos* 30(4): 657–673.

Siebert, H. (1981). Nature as life support system: Renewable resources and environmental disruption. *Journal of Economics* 42(2): 133–142.

Siebert, H. (1983). *Ökonomische Theorie natürlicher Ressourcen.* Tübingen: Mohr Siebeck.

Siebert, H. (1996). Trade policy and environmental protection. *The World Economy* 19(5/Supplement): 183–194.

Siebert, H. (2000). *Außenwirtschaft.* Seventh revised edition. Stuttgart: Lucius & Lucius.

Siebert, H. (2005). *Economics of the environment.* Sixth edition. Berlin: Springer.

Silard, S.A. (1995). The Global Environment Facility: A new development in international law and organization. *The George Washington Journal of International Law an Economics* 28(3): 607–654.

Simpson, R.D., and R.A. Sedjo (1994). Commercialization of indigenous genetic resources. *Contemporary Policy* 12(4): 34–44.

Simpson, R.D., and R.A. Sedjo (1996). Valuation of biodiversity for use in new product research in a model of sequential search. Discussion Paper 96-27. Resources for the Future, Washington, D.C.

Simpson, R.D., R.A. Sedjo, and J.W. Reid (1996). Valuing biodiversity for use in pharmaceutical research. *Journal of Political Economy* 104(1): 163–185.

SIPA (School of International and Public Affairs) (1999). Access to genetic resources: An evaluation of the development and implementation of recent regulation and access agreements. Environmental Policy Studies. Working Paper #4. Columbia University, New York.

Sjöberg, H. (1999). Restructuring the Global Environment Facility. GEF Working Paper 13. GEF, Washington, D.C.

Smale, M. (1997). The Green Revolution and wheat genetic diversity: Some unfounded assumptions. *World Development* 25(8): 1257–1269.

Smale, M., J. Hartell, P.W. Heisey, and B. Senhauer (1998). The contribution of genetic resources and diversity to wheat production in the Punjab of Pakistan. *American Journal of Agricultural Economics* 80(3): 482–493.

Small, A.A. (1998). The market for genetic resources: The role of research and development in the valuation and conservation of biological intellectual capital. University of California Berkeley. Dissertation. Berkeley, CA.

Smith, D.L., S.A. Levin, and R. Laxminarayan (2005). Strategic interactions in multi-institutional epidemics of antibiotic resistance. Proceeding of the National Academy of Sciences (PNAS) 102(8): 3153–3158.

Smith, R.J., R.D.J. Muir, M.J. Walpole, A. Balmford, and N. Leader-Williams (2003). Governance and the loss of biodiversity. *Nature* 426: 67–70.

Smith, S.E. (2000). What is an environmental fund, and when is it the right tool for conservation? In R.A. Norris (ed.), *The IPG Handbook on Environmental Funds: A resource book for design and operation of environmental funds.* New York City: IPG (Inter-Agency Planning Group on Environmental Funds).

Solow, A., S. Polasky, and J. Broadus (1993). On the measurement of biological diversity. *Journal of Environmental Economics and Management* 24(1): 60–68.

Soulé, M.E., and M.A. Sanjayan (1998). Conservation Targets: Do They Help? *Science* 279: 2060–2061.

Spalding, M. (2002). The World Heritage List: The best of all worlds? *Parks* 12(3): 50–57.

Srinivasan, C.S. (2003). Exploring the feasibility of Farmers' Rights. *Development Policy Review* 21(4): 419–447.

Srinivasan, C.S., and C. Thirtle (2000). Understanding the emergence of terminator technologies. *Journal of International Development* 12(8): 1147–1158.

Srinivasan, C.S., and C. Thirtle (2003). Potential economic impacts of terminator technologies: Policy implications for developing countries. *Environment and Development Economics* 8(1): 187–205.

Stähler, F. (1992). The international management of biodiversity. Kiel Working Paper No. 529. Kiel Institute for World Economics, Kiel.

Stähler, F. (1994). Biological diversity: The international management of genetic resources and its impact on biotechnology. *Ecological Economics* 11(3): 227–236.

Stähler, F. (1996). On international compensations for environmental stocks. *Environmental and Resource Economics* 8(1): 1–13.

Stegemann, K. (2000). The integration of intellectual property rights into the WTO system. *The World Economy* 23(9): 1237–1267.

Stevenson, G.G. (1991). *Common property economics: A general theory and land use application.* Cambridge: Cambridge University Press.

Stevenson, M. (2002). China, Brazil, India, 9 other nations form alliance against biopiracy. Associated Press. February 18, 2002.

Stoll, P.-T., S. Franck, S. Reyes-Knoche, and F. Höhne (2004). Armutsbekämpfung und Zugang zu genetischen Ressourcen. Externe Expertise for das WBGU-Hauptgutachten „Welt im Wandel: Armutsbekämpfung durch Umweltpolitik." Berlin: WBGU (Wissenschaftlicher Beirat der Bundesregierung Globale Umweltveränderungen/German Advisory Council on Global Change).

Ströbele, W.J. (1991). Institutional arrangements for transfrontier river pollution. In H. Siebert (ed.), *Environmental scarcity: The international dimension.* Tübingen: Mohr Siebeck.

Subramanian, A. (1992). Genetic resources, biodiversity and environmental protection. *Journal of World Trade* 26(5): 105–109.

Swanson, T. (1994). The economics of extinction revisited and revised: A generalised framework for the analysis of the problems of endangered species and biodiversity losses. *Oxford Economic Papers* 46(Supplement): 800–821.

Swanson, T. (1995a). The international regulation of biodiversity decline: Optimal policy and evolutionary product. In C. Perrings et al. (eds.) *Biodiversity loss: Economic and ecological issues.* Cambridge: Cambridge University Press.

Swanson, T. (1995b). The appropriation of evolution's values: An institutional analysis of intellectual property regimes and biodiversity conservation. In T. Swanson (ed.), *Intellectual property rights and biodiversity conservation: An interdisciplinary analyses of the values of medicinal plants.* Cambridge: Cambridge University Press.

Swanson, T. (1996). The reliance of northern economies on southern biodiversity: Biodiversity as information. *Ecological Economics* 17(1): 1–8.

Swanson, T. (1997). Global action for biodiversity. An international framework for implementing the Convention on Biological Diversity. London: Earthscan.

Swanson, T. (1999). Why is there a biodiversity convention? The international interest in centralized development planning. *International Affairs* 75(2): 307–331.

Swanson, T. (2002). Biotechnologies and developing countries: How will the anticipated industrial change in agriculture affect developing countries. In T. Swanson (ed.), *Biotechnology, agriculture and the developing world: The distributional implication of technological change.* Cheltenham: Edward Elgar.

Swanson, T., and T. Goeschl (2000). Property rights issues involving plant genetic resources: Implications of ownership for economic efficiency. *Ecological Economics* 32(1): 75–92.

Swanson, T., and T. Goeschl (2002). The impact of GURTs: Agricultural R&D and appropriation mechanism. In T. Swanson (ed.), *Biotechnology, agriculture and the developing world: The distributional implication of technological change.* Cheltenham: Edward Elgar.

Swanson, T., and A. Kontoleon (2000). Why did the protected areas fail the giant panda? The economics of conserving endangered species in developing countries. *World Economics* 1(4): 135–148.

Swanson, T., and R.A. Luxmore (1997). Industrial reliance on biodiversity. A Darwin Initiative Project. *WCMC Biodiversity Series* No.7. World Conservation Monitoring Centre, Cambridge.

Tacconi, L., and J. Bennett (1995). Economic implications of intergenerational equity for biodiversity conservation. *Ecological Economics* 12(3): 209–223.

ten Kate, K. (2002). Global genetic resources: Science and the convention on biological diversity. *Science* 295: 2371–2372.

ten Kate, K., and S.A. Laird (1999). *The commercial use of biodiversity: Access to genetic resources and benefit sharing.* London: Earthscan.

ten Kate, K., and S.A. Laird (2000). Biodiversity and business: Coming to terms with the 'Grand Bargain'. *International Affairs* 76(2): 241–264.

Thomas, C.D., A. Cameron, R.E. Green, M. Bakkenes, L.J. Beaumont, Y.C. Collingham, B.F.N. Erasmus, M. Ferreira de Siqueira, A. Grainger, L. Hannah, L. Hughes, B. Huntley, A.S. van Jaarsveld, G.F. Midgley, L. Miles, M.A. Ortega-Huerta, A.T. Peterson, O.L. Phillips, and S.E. Williams (2004). Extinction risk from climate change. *Nature* 427: 145–148.

Thrupp, L.A. (2000). Linking agricultural biodiversity and food security: The valuable role of agrobiodiversity for sustainable agriculture. *International Affairs* 76(2): 283–297.

Tilman, D. (1999). The ecological consequences of changes in biodiversity: A search for general principles. *Ecology* 80(5): 1455–1474.

Tilman, D. (2000). Causes, consequences and ethics of biodiversity. *Science* 405: 208–211.

Tilman, D., S. Polasky, and C. Lehman (2005). Diversity, productivity and temporal stability in the economies of humans and nature. *Journal of Environmental Economics and Management* 49(3): 405–426.

Timpson, S. (2000). Working with the GEF Small Grant Programme. In R.A. Norris (ed.), *The IPG Handbook on Environmental Funds A resource book for design and operation of environmental funds.* New York City: IPG (Inter-Agency Planning Group on Environmental Funds).

Turner, R.K., J. Paavola, P. Cooper, and S. Farber (2003). Valuing nature: Lessons learned and future research directions. *Ecological Economics* 46(3): 493–510.

Ulph A., and D. Ulph (1997). Global warming, irreversibility and learning. *Economic Journal* 107(442): 636–650.

UNDP (United Nations Development Programme) (1999). *Human development report 1999.* New York.

UNDP (United Nations Development Programme) (2005). Biodiversity after Johannesburg: The critical role of biodiversity and ecosystem services in achieving the United Nations Millennium Goals. Available at <http://www.iisd.ca/sd/sdund/sdvol81num1e.html >, accessed on December 30, 2005.

UNEP (United Nations Environment Programme) (2002). Replenishment of Global Environment Facility Seen as Major Success. for Johannesburg Summit Process. Johannesburg Summit for Sustainable Development. Press Release. New York, 14 August.

UNEP/CBD (United Nations Environment Programme/Convention on Biological Diversity) (1996a). The availability of additional financial resources. Note by the Executive Secretary. UNEP/CBD/COP/3/37. Buenos Aires.

UNEP/CBD (United Nations Environment Programme/Convention on Biological Diversity) (1996b). Ways and means to promote and facilitate access, and transfer, and development of technology, including biotechnology. Subsidiary Body on Science, Technical and Technological Advice. Second Meeting. Montreal, 2–6 August 1996. UNEP/CBD/SBSTTA/2/6.

UNEP/CBD (United Nations Environment Programme/Convention on Biological Diversity) (1998). Synthesis of case-studies on benefit-sharing. Conference of the Parties to the Convention of Biological Diversity. Fourth Meeting. Bratislava, 4–15 May 1998. UNEP/CBD/COP/4/Inf.7.

UNEP/CBD (United Nations Environment Programme/Convention on Biological Diversity) (2002a). Report of the Global Environmental Facility. Addendum: Second Overall Performance Study of the Global Environmental Facility. UNEP/CBD/COP/6/9/Add.1. March 11, 2002. The Hague.

UNEP/CBD (United Nations Environment Programme/Convention on Biological Diversity) (2002b). Thematic programmes of work: Progress reports on implementation: Agricultural biological diversity. Potential impacts of genetic use restriction technologies (GURTs) on agricultural biodiversity and agricultural production systems. Note by the Executive Secretary. Conference of the Parties to the Convention of Biological Diversity. Sixth Meeting. The Hague, 7–19 April 2002. UNEP/CBD/COP/6/INF/1/Rev.1.

UNEP/CBD (United Nations Environment Programme/Convention on Biological Diversity) (2005). Ad hoc open-ended Working Group on Access and Benefit-Sharing: Analysis of existing national, regional and international legal instruments relating to access and benefit sharing and experience gained in their implementation, including identification of gaps. Bangkok, 14–18 February 2005. UNEP/CBD/WG-ABS/3/2.

UPOV (Union for the Protection of New Varieties of Plants) (2005). States party to the International Convention for the Protection of New Varieties of Plants. Status on November 9, 2004. Available at <http://www.upov.int/en/about/members/index.htm>, accessed on March 15, 2005.

van der Veen, A., and H.S. Otter (2001). Land use changes in regional economic theory. *Environmental Modeling and Assessment* 6(2): 145–150.

van Kooten, G.C., and E.H. Bulte (2000). *The economics of nature: Managing biological assets.* Oxford: Blackwell Publishing.

Van Soest, D., and R. Lensink (2000). Foreign transfers and tropical deforestation: What terms of conditionality? *American Journal of Agricultural Economics* 82(2): 389–399.

Varian, H. (1992). *Microeconomic analysis.* 3rd edition. New York: Norton.

Varian, H. (1999). Markets for information goods. IMES discussion paper series E 99:9. Institute for Monetary and Economic Studies, Tokyo.

Vaughan, D. (1999). Tourism and biodiversity: A convergence of interests? *International Affairs* 76(2): 283–297.

VFA (Verband Forschender Arzneimittelhersteller e.V.) (2004). Statistics 2003. Die Arzneimittelindustrie in Deutschland. Berlin.

Virchow, D. (1999a). Conservation of plant genetic resources for food and agriculture: Main actors and the costs to bear. *International Journal of Social Economics* 26(7/8/9): 1144–1161.

Virchow, D. (1999b). Conservation of genetic resources: Costs and implications for a sustainable utilization of plant genetic resources for food and agriculture. Berlin: Springer.

Visser, B., D. Eaton, N. Louwaars, and J. Engels. (2000). Transaction costs of germplasm exchange under bilateral agreements. Paper presented at the Global Forum on Agricultural Research (GFAR), 21–23 May, Dresden, Germany.

von Amsberg, J. (1995). Excessive environmental risks: An intergenerational market failure. *European Economic Review* 39(1): 1447–1464.

Walden, I. (1995). Preserving biodiversity: The role of property rights. In T. Swanson (ed.), *Intellectual property rights and biodiversity conservation: An interdisciplinary analyses of the values of medicinal plants.* Cambridge: Cambridge University Press.

Warr, P.G. (1983). The private provision of public goods is independent of the distribution of income. *Economics Letters* 13(2–3): 207–211.

WBGU (Wissenschaftlicher Beirat der Bundesregierung Globale Umweltveränderungen/ German Advisory Council on Global Change) (2000). World in transition: Conservation and sustainable use of the biosphere. Annual Report 1999. London: Earthscan.

WBGU (Wissenschaftlicher Beirat der Bundesregierung Globale Umweltveränderungen/ German Advisory Council on Global Change) (2002). Charging the use of global commons. Special Report 2002. Berlin.

WBGU (Wissenschaftlicher Beirat der Bundesregierung Globale Umweltveränderungen/ German Advisory Council on Global Change) (2005). World in transition four: Fighting poverty through environmental policy. Annual Report 2004. London: Earthscan.

WCPA (World Commission on Protected Areas) (2000). Financing protected areas: Guidelines for protected area managers. Financing Protected Areas Task Force of the World Commission on Protected Areas (WCPA) of IUCN, in collaboration with the Economics Unit of IUCN. Gland: IUCN.

Weber, M. (2004). Assessing the effectiveness of tradable landuse rights for biodiversity conservation: An application to Canada's boreal mixed wood forest. Nota di lavoro 2004.29. Fondazione Eni Enrico Mattei, Milan.

Weisbrod, B.A. (1964). Collective-consumption services of individual-consumption goods. *The Quarterly Journal of Economics* 78(3): 471–477.

Weitzman, M. (1974). Prices vs. quantities. *Review of Economic Studies* 41(4): 477–491.

Weitzman, M. (1979). Optimal search for the best alternative. *Econometrica* 47(3): 641–654.

Weitzman, M. (1992). On diversity. *The Quarterly Journal of Economics* 107(2): 363–405.

Weitzman, M. (1993). What to preserve? An application of diversity theory to crane conservation. *The Quarterly Journal of Economics* 108(1): 157–183

Weitzman, M. (1998). The Noah's ark problem. *Econometrica* 66(6): 1279–1298.

Weitzman, M. (2000). Economic profitability versus ecological entropy. *The Quarterly Journal of Economics* 115(1): 237–263.

WHC (World Heritage Convention) (2002): The World Heritage Fund. Available at <http://whc.unesco.org/ab_fund.htm>, accessed on July 15, 2005.

Williamson, O.E. (1971). The vertical integration of production: Market failure consideration. *American Economic Review* 61(2): 112–123.

Williamson, O.E. (1998). Transaction cost economics: How it works; where it is headed. *De Economist* 146(1): 23–58.

Wolfensohn, J.D. (2004). Für einen besseren Umweltschutz. *Sueddeutsche Zeitung* June 8(2004):2.

Wolfrum, R., G. Klepper, P. Stoll, and S. Franck (2001). Die weitere Umsetzung der CBD: Analyse der Zusammenhänge mit Fragen des geistigen Eigentums und mit dem internationalen System zum Schutzes geistiger Eigentumsrechte. UFOPLAN 1998: Analyse der Ergebnisse der 4. Vertragsstaatenkonferenz der Biodiversitätskonvention unter besonderer Berücksichtigung der Fragen des geistigen Eigentums. Berlin: Umweltbundesamt.

World Bank (1995). Mainstreaming biodiversity in development: A World Bank assistance strategy for implementing the Convention on Biological Diversity. Environmentally Sustainable Development (ESD). Washington, D.C.

World Bank (2003). Cornerstones for conservation: World Bank assistance for protected areas. 1988–2003. Washington, D.C.

World Bank (2004). World Development Indicators 2004: Data Set. Washington, D.C.

World Bank (2005). Biodiversity. Available at <http://lnweb18.worldbank.org/ESSD/envext.nsf/48ByDocName/Biodiversity>, accessed on December 30, 2005.

WRI (World Resource Institute) (2000). A guide to world resources 2000–2001. People and ecosystems: The fraying web of life. United Nations Development Programme, United Nations Environment Programme, World Bank, and World Resource Institute, Washington, D.C.

WRI (World Resources Institute) (2003). Protected areas. EarthTrends data tables: Biodiversity and protected areas. Washington, D.C. Available at <http://earthtrends.wri.org/>, accessed on January 7, 2004.

WTO/CTE (World Trade Organization / Committee on Trade and Environment) (1995). Environment and TRIPS. WT/CTE/W/8. Geneva.

Wunder, S. (2000). Ecotourism and economic incentives. *Ecological Economics* 32(3): 465–479.

WWF International (World Wildlife Fund International) (1999). Living Planet Report 1999. Gland.

Young, M.D. (1992). Sustainable investment and resource use: Equity, environmental integrity and economic efficiency. Man And The Biosphere Series Volume 9. Paris: UNESCO.

Zilberman, D., H. Ameden, G. Graff, and M. Quaim (2004). Agricultural biotechnology: Productivity, biodiversity, and intellectual property rights. *Journal of Agricultural & Food Industrial Organization* 2(3).