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Economic Instruments for Conservation Policies in Federal Systems

Irene Ring

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Economic Instruments for Conservation Policies in Federal Systems

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Gutachter: Prof. Dr. Thomas Lenk, Institut für Öffentliche Finanzen und Public Management / Finanzwissenschaft, Universität Leipzig, Germany

Prof. Dr. John M. Gowdy, Department of Economics, Rensselaer Polytechnic Institute, Troy, N.Y., USA

Prof. Dr. Arild Vatn, Department of International Environment and Development Studies, Norwegian University of Life Sciences (UMB), Ås, Norway

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Preface

One of the early articles included in this habilitation thesis concludes: “Research at the interface of implementing the sustainability concept, conservation and environmental policies, and the economic theory of federalism is still more or less in its infancy” (see chapter 2, p. 84). Just a few years later, global environmental change has entered our daily lives and made the headlines in newspaper and on TV news. New approaches to public finance are needed to help us design innovative policies capable of responding to the global environmental challenges we face. A “New Public Finance” – as Kaul and Conceição (2006)[1] frame it – has emerged only very recently that is responding to global challenges, addressing economic, environmental and social problems alike. Global climate change and the loss of biodiversity worldwide have come onto the political agenda, prompting the need for new perspectives and solutions.

With this thesis, I would like to make a contribution to this internationally thriving field of research: a public finance that truly integrates environmental and conservation policies, builds on sustainability science, and encompasses a governance perspective addressing local, national and global issues, as well as public and private sector interaction.

This habilitation thesis presents the key results of my research at the Helmholtz Centre for Environmental Research – UFZ. First of all, I would like to express sincere thanks my primary supervisor (*Erstgutachter*) Prof. Dr. Thomas Lenk, University of Leipzig, for his ongoing support over the years and for the ease with which mutual cooperation was able to flourish, both in teaching and in research activities in public finance, intergovernmental fiscal relations and ecological economics. I am indebted to Prof. Dr. Bernd Hansjürgens, UFZ and Martin Luther University Halle-Wittenberg, who granted me the freedom to concentrate on this thesis during critical work phases and who also provided substantial encouragement and valuable comments. Further thanks go to Prof. Dr. Erik Gawel, UFZ and University of Leipzig, for his helpful comments on the introduction, and my colleagues at the Department of Economics at UFZ for their encouraging team spirit. Special thanks go to my partner Dr. Herwig Unnerstall for many valuable discussions on the topic and sustained support. Last but not least, I would like to thank Sabine Linke[†] for assisting me with the formalities of putting this thesis together.

Leipzig, July 2011 Irene Ring

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[1] Kaul, I., Conceição, P. (2006). The New Public Finance: Responding to Global Challenges. Oxford University Press, New York and Oxford.

Editorial notes

This cumulative habilitation is based on previously published articles. Chapters 2–9 consist of seven journal articles and one book chapter, which appear largely in the form they were originally published. Each chapter contains an abstract (and in some cases a summary), keywords (where these were included in the original publication) and a separate bibliography. In addition, each chapter is prefaced with a newly written brief introduction, explaining the history of the original publication.

In order to achieve a common format across the various chapters of this habilitation thesis, some of the articles have been slightly edited. Abbreviations for the German federal states (*Länder*) have been harmonised across respective tables in chapters 2, 3 and 4. Where references appeared in footnotes in the original article (chapters 3 and 9), they have now been included in a references section at the end of the chapter. Endnotes (chapter 8) have been transformed into footnotes. All photographs have been treated as figures and the "Excursus" (chapter 6) presented in a box. The title and all figures appearing in the first German publication (chapter 3) have been complemented by a title and figure captions in English language as was already the case with the other German original publication (chapter 7). Wherever apparant, references to works in press and (in some cases) to internet addresses (URLs) have been updated in the references sections. Acknowledgements of individual articles now appear towards the end of each chapter rather than in footnotes (chapters 3 and 7). Chapter 8 included cross-references to other chapters in the same book, and these have therefore been changed to proper citations.

Permission to reprint selections from the following sources is gratefully acknowledged:

Chapter 2: Ecological public functions and fiscal equalisation at the local level in Germany. Published in 2002 in Ecological Economics 42(3): 415–427, Elsevier Science, The Netherlands.

Chapter 3: Ökologische Aufgaben und ihre Berücksichtigung im kommunalen Finanzausgleich. Published in 2001 in H. Spehl and M. Held (Eds.), Vom Wert der Vielfalt. Zeitschrift für angewandte Umweltforschung, Sonderheft 13: 236–249, reproduced with kind permission of Analytica Verlag, Germany.

Chapter 4: Compensating municipalities for protected areas. Fiscal transfers for biodiversity conservation in Saxony, Germany. Published in 2008 in GAIA 17/S1: 143–151, reproduced with kind permission of oekom Verlag, Germany.

Chapter 5: Integrating local ecological services into intergovernmental fiscal transfers: The case of the ecological ICMS in Brazil. Published in 2008 in Land Use Policy 25(4): 485–497, Elsevier Science, The Netherlands.

Chapter 6: Biodiversity. Emerging issues for linking natural and social sciences. Published in 2003 with A. Jentsch, H. Wittmer, K. Jax, and K. Henle in GAIA 12(2): 121–128, reproduced with kind permission of oekom Verlag, Germany.

Chapter 7: Naturschutz in der föderalen Aufgabenteilung: Zur Notwendigkeit einer Bundeskompetenz aus ökonomischer Perspektive. Published in 2004 in Natur und Landschaft 79(11): 494–500, W. Kohlhammer Verlag, Germany. Photo credits: Dirk Nette (Figure 7.1), Naturfoto Pretscher (Figure 7.4).

Chapter 8: Biodiversity Governance: Adjusting local costs and global benefits. Published in 2008 in T. Sikor (Ed.), Public and Private in Natural Resource Governance: A False Dichotomy? Earthscan, London, pp. 107–126, reproduced with kind permission of Earthscan Ltd www.earthscan.co.uk.

Chapter 9: Protected species in conflict with fisheries: The interplay between European and national regulation. Published in 2006 with J. Similä, R. Thum and R. Varjopuro in Journal of European Environmental and Planning Law 3(5): 432–445, reproduced with kind permission of Koninklijke Brill NV, The Netherlands.

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List of abbreviations

Abl.	Ablage
ACP	African, Caribbean and Pacific
APAs	Environmental Protection Areas
Art.	Artikel
Aufl.	Auflage
BfN	Bundesamt für Naturschutz
BMU	Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit
bzw.	beziehungsweise
CAP	Common Agricultural Policy
CDU	Christlich Demokratische Union
cf.	confer
CFP	Common Fisheries Policy
CO	Colorado
CO_2	Kohlendioxid
COM	Commission of the European Communities
COPAM	O Conselho de Política Ambiental
CPRH	State Agency of the Environment of Pernambuco
CSU	Christlich Soziale Union
CUs	Conservation Units
DCBIO	Diretoria de Conservação da Biodiversidade
DG	Direction General
d.h.	das heißt
DIW	Deutsches Institut für Wirtschaftsforschung
DM	Deutsche Mark
DNR	Deutscher Naturschutzring e.V.
EAGGF	European Agricultural Guidance and Guarantee Fund
EC	European Communities
ECJ	European Court of Justice
Ed. / Eds.	Editor / Editors

EEA	European Environment Agency
EEC	European Economic Community
e.g.	exempli gratia
EGV	Europäischer Gemeinschaftsvertrag
EPA-SAB	U.S. Environmental Protection Agency – Science Advisory Board
Eq.	Equation
ERDF	European Regional Development Fund
ESF	European Social Fund
et al.	et alii
etc.	et cetera
EU	Europäische Union
EUV	Vertrag über die Europäische Union
EWG	Europäische Wirtschaftsgemeinschaft
f. / ff.	following / folgende
FAG	Finanzausgleichsgesetz
FAO	Food and Agriculture Organisation
FFH	Fauna Flora Habitat-Richtlinie
FIAF	Finanzinstrument für die Ausrichtung der Fischerei
FIFG	Financial Instrument for Fisheries Guidance
FiFo	Finanzwissenschaftliches Forschungsinstitut an der Universität zu Köln
FP	Framework Programme
FRAP	Framework for biodiversity Reconciliation Action Plan
GBl.	Gesetzblatt
GDP	Gross Domestic Product
GFG	Gemeindefinanzierungsgesetz
GG	Grundgesetz
GIS	Geographical Information System / Geographisches Informationssystem
GTZ	Gesellschaft für Technische Zusammenarbeit
GVBl.	Gesetz- und Verordnungsblatt
GV NRW	Gesetz- und Verordnungsblatt für das Land Nordrhein-Westfalen
GVOBl.	Gesetz- und Verordnungsblatt
H.	Heft
ha	Hektar

Hrsg.	Herausgeber
i.A.	im Auftrag
IAP	Instituto Ambiental do Paraná
IBAMA	Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis
Ibid	ibidem / ebenda
ICES	International Council for the Exploration of the Sea
ICMS-E	Imposto sobre Circulação de Mercadorias e Serviços-Ecológico
i.d.F.	in der Fassung
i.d.F.d.Bek.v.	in der Fassung der Bekanntmachung vom
i.e.	id est, das heißt
IFM	Innovative Fisheries Management (Aarlborg University Research Centre)
inter alia	unter anderem
INTERREG	Gemeinschaftsinitiative des Europäischen Fonds für regionale Entwicklung zur Förderung der Zusammenarbeit zwischen den Regionen der Europäischen Union
IPCC	Intergovernmental Panel on Climate Change
i.V.m.	in Verbindung mit
Jg.	Jahrgang
Jr.	Junior
KFA	Kommunaler Finanzausgleich
KFAG	Kommunalfinanzausgleichsgesetz
km²	Square kilometre / Quadratkilometer
LFAG	Landesfinanzausgleichsgesetz
MA	Massachusetts
MA	Millennium Ecosystem Assessment
MCF	Municipal Conservation Factor
MD	Maryland
MEA	Millennium Ecosystem Assessment
MMA	Ministério do Meio Ambiente / Ministry of Environment (Brazil)
MoAF	Ministry of Agriculture and Forestry (Finland)
Mrd.	Milliarde
MT	Montana
NAF	Nordisk Arbeitsgruppe for Fiskeriforskning / Nordic Working Group for Fishery Research
NAK	Naturschutz und Erhalt der Kulturlandschaft

NFAG	Niedersächsisches Gesetz über den Finanzausgleich
NGO	Non-Governmental Organisation
No.	Number / numéro
Nr.	Nummer
NWRC	National Wildlife Research Center
Oct.	October
OECD	Organisation for Economic Co-operation and Development
OJ	Official Journal of the European Union
p. / pp.	page / pages
PERC	Property and Environment Research Center
PES	Payments for Environmental Services
R $	Brazilian Real
RL-Nr.	Richtlinien-Nummer
RPPNs	Reserva Particular do Patrimônio Natural / Private Natural Patrimony Reserve
S.	Seite
s.	siehe
SAC	Special Area of Conservation
SächsGVBl	Sächsisches Gesetz- und Verordnungsblatt
SBF	Secretaria de Biodiversidade e Florestas
SCU	Science Communication Unit
SEGES	Secretaria de Gestão Público
SEMARH	Secretaria do Meio Ambiente e dos Recursos Hídricos
SEPLAN	Secretaria do Planejamento e Meio Ambiente
SMF	Sächsisches Staatsministerium der Finanzen
SPA	Special Protection Area
SRU	Sachverständigenrat für Umweltfragen
TEEB	The Economics of Ecosystems and Biodiversity
ThürFAG	Thüringer Finanzausgleichsgesetz
Tz	Textziffer
u.a.	unter anderem
u.a.O.	und anderen Orts
UC	Unidade de Conservação

UFZ	Helmholtz-Zentrum für Umweltforschung – UFZ (früher: UFZ – Umweltforschungszentrum Leipzig-Halle GmbH)
UK	United Kingdom
UL	Förderprogramm Umweltgerechte Landwirtschaft im Freistaat Sachsen
UNEP	United Nations Environment Programme
unter Mitarb.	unter Mitarbeit
UPI	Umwelt- und Prognose-Institut Heidelberg e.V.
U.S.	United States
US $	US-Dollar
USA	United States of America
UTR	Umwelt- und Technikrecht
v.	vom
VG	Verwaltungsgrenze
vgl.	vergleiche
VO	Verordnung
WBGU	Wissenschaftlicher Beirat der Bundesregierung Globale Umweltveränderungen
WCED	World Commission on Environment and Development
WHI	Walter Hallstein Institut (für Europäisches Verfassungsrecht)
WWF	World Wildlife Fund (heute: World Wide Fund for nature)
ZAU	Zeitschrift für angewandte Umweltforschung
z.B.	zum Beispiel
z.T.	zum Teil
zul. geänd. d. G. v.	zuletzt geändert durch Gesetz vom

Part I:

Introduction

1 Economic instruments for conservation policies – State of the art and the way forward

This habilitation thesis consists of seven published journal articles and one published book chapter. The title “Economic Instruments for Conservation Policies in Federal Systems” captures the common theme of the following chapters, which address three major areas of economic research:

1) With its core papers on intergovernmental fiscal transfers in environmental and conservation policies, the habilitation thesis breaks new ground at the interface of environmental economics and public finance, both in theoretical and applied research;
2) Building on methodological pluralism and interdisciplinary research, it provides new insights for ecological economics related to integrative biodiversity research, policy analysis, and ecological fiscal transfers;
3) Through its orientation towards policy relevance and real-world institutions, it contributes to governance research and institutional economics.

The structure of the thesis is as follows:

Part I, consisting of chapter 1, introduces the overarching theme of environmental policy instruments and discusses the current use of and prospects for economic instruments in conservation policies. A number of research gaps are identified which are addressed in the subsequent chapters.

Part II, consisting of chapters 2–5, encompasses four papers focused on a single type of policy instrument: intergovernmental fiscal transfers. Although well documented in public finance literature, intergovernmental fiscal transfers remain a somewhat neglected instrument in environmental policy. Despite being well suited to address the spillover benefits that often accrue with conservation policies, there is scant research literature on ecological fiscal transfers compared to other economic instruments such as environmental taxes or tradable permits. In fact, very few countries make practical use of them to achieve conservation objectives. Thus intergovernmental fiscal transfers are an innovative instrument in conservation policies in particular, so that advances in both theory and applied research may prove especially beneficial here. It is against this background that Part II presents the following contributions:

- the need for integrating nature conservation issues into intergovernmental fiscal transfers is substantiated from an economic and ecological perspective and, building on the economic theory of federalism and insights from sustainability science, some initial proposals are presented for their integration (chapters 2 and 3);
- suitable indicators are developed for integrating nature conservation into intergovernmental fiscal transfers and, by way of example, the fiscal impacts of using these indi-

cators are modelled in a spatially explicit way for fiscal equalisation at the local level in Saxony, a federal state (*Land*) in Germany (chapter 4);

- finally, the design and implementation of existing ecological fiscal transfers to the local level in Brazil are investigated, along with their fiscal and ecological effects, and the potential for their transfer to other federal countries is examined (chapter 5).

Part III, consisting of chapters 6–9, combines a number of articles in integrative biodiversity research and applied biodiversity governance, themes that are often neglected in the economic analysis of environmental policy instruments. However, when implementing policy instruments in societal settings, interdisciplinary research bridging the natural and social sciences is as much a prerequisite as policy-relevant research that responds to the needs of decision makers and other stakeholders. Both policy design and policy evaluation yield the best outcomes when they involve ecologists, economists, legal and other social scientists, as this ensures that consideration is given to ecological effectiveness, economic efficiency, administrative feasibility, social acceptance, and perception by stakeholders. Policy-relevant research also responds to current societal developments and prospective changes in legislation which may provide windows of opportunity to propose new instruments. Meanwhile, sound empirical research and case study design are indispensable in making concrete policy recommendations, taking into account existing formal and informal institutions. In this context,

- a sound scientific basis is established for integrative biodiversity research and policy drawing on ecological, economic as well as broader institutional and social science perspectives (chapter 6);

- theoretical arguments building on the economic theory of federalism are developed to provide economic justification for an adequate distribution of conservation competencies in federal systems, serving also as a robust basis for policy recommendations in the course of modernisation of the federal structure in Germany (chapter 7);

- the economic theory of federalism is linked at a theoretical level to an institutional and governance-oriented perspective of conservation policies, followed by an examination of the design and implementation of ecological fiscal transfers in Brazil in multi-level and multi-actor perspective (chapter 8); and, finally,

- economic, legal and anthropological expertise is combined to analyse the existing legal and institutional framework in Europe; in addition, building on theoretical insights from different disciplinary backgrounds and enriched by empirical research from regional case studies in two European member states, recommendations for better coordination of conservation policies in federal systems are proposed (chapter 9).

The remainder of this introduction is organised as follows: The first section presents a concise analysis of environmental policy instruments, with an emphasis on economic instruments (section 1.1). A special focus is placed on the role of the state in environmental and conservation policies[1]. Fiscal instruments are of particular relevance to public decision makers, and in this regard the concept of ecological fiscal reform is introduced (subsection 1.1.2). With regard to the field of biodiversity and nature conservation, a number of gaps in the design and use of fiscal instruments can be identified (subsection 1.1.3) which, step by step, are filled in the course of the chapters that follow. The first section closes with some comments on the challenges of implementing policy instruments in federal systems, followed by the introduction of the concept of biodiversity governance (subsection 1.1.4). The subsequent two sections of chapter 1 present the background to and main results of the published articles constituting this thesis (sections 1.2 and 1.3). The articles are grouped into two main sections, mirroring the basic structure of the thesis.

1.1 Economic instruments in environmental and conservation policies

1.1.1 Economic instruments in environmental policy

As Oates (1999a, p. xiii) puts it, "economics has three basic and important messages for environmental protection":

1) an unfettered market system generates excessive pollution. Hence economic reasoning argues "for the need of public intervention in the form of environmental regulation" (ibid.);
2) economics offers guidance for setting environmental quality standards, based on cost-benefit-analysis. However, economic valuation of benefits and costs builds on marginal analysis and one has to be aware of the limits of its application, from both an ecological and a socio-cultural perspective as Vatn and Bromley (1994), Turner et al. (2003), Balmford et al. (2008), EPA-SAB (2009) and Ring et al. (2010b), among others, point out;
3) once environmental objectives have been determined – by economic analysis or by political decision making following a standards and pricing approach (Baumol and Oates 1988) –, economic analysis provides insights into the design of policy instruments to achieve these targets "in the most effective and least-cost way" (Oates 1999a, p. xiv).

The last of these messages is ultimately about the choice of policy instruments and their design and implementation, which is the focus of this habilitation thesis. Early environmental legislation generally used so-called command-and-control approaches, often prescribing definite pollution control techniques to polluters. Economic analysis has shown in many studies

[1] Although the term environmental policy is sometimes used as an overarching term to include conservation issues, here it is distinguished from the term conservation policy. Conservation policies include nature conservation, landscape protection as well as the conservation and sustainable use of biodiversity. In short, conservation policies involve biota, i.e. living organisms and their associated (eco-)systems. In contrast, environmental policy deals with the abiotic aspects of environmental protection and degradation. This distinction is commonly made in environmental and conservation laws as well as in the assignment of competencies to expert authorities in many countries (e.g., in Germany: Environmental Protection Agency / *Umweltbundesamt* and Federal Agency for Nature Conservation / *Bundesamt für Naturschutz)*.

that this approach entails high costs for the polluter and for society. Economic argumentation is in favour of incentive-based policy instruments, i.e. environmental taxes or tradable permits that introduce a price as a market signal to polluters. The price signal leaves it to the polluter, in consideration of the cost structure of their business, to decide how to adapt: whether to pay for continued pollution or to adopt emission control technologies to avoid polluting the environment (Hansmeyer and Schneider 1990; Oates 1999a).

Building on the economic theory of public goods and the related theory of externalities as cornerstones, environmental economics literature and textbooks over the last 30 to 40 years have consistently presented the basic structure of environmental problems as consisting in a firm polluting the environment through air or water emissions (e.g., Baumol and Oates 1975, 1988; Siebert 1978; Fisher 1981; Endres 1985; Turner et al. 1993; Cansier 1996; Feess 2007). In environmental policy analysis, the economist's task is then to find the most effective and least-cost way to achieve a certain environmental standard. Economic research focused initially on improving the effectiveness and efficiency of single environmental policy instruments (e.g., Endres 1985). Subsequently, investigations turned towards evaluating and improving real-world instrument mixes, given that legal, planning, fiscal, and other economic instruments exist in parallel (Gawel 1991; Sterner 2003; OECD 2007; Lehmann 2010).

Early discussions among economists were related to whether price-driven environmental taxes and charges or quantity-related tradable permits would be more efficient to achieve a set target (e.g., Weitzman 1974; Siebert 1976; Bonus 1981, 1998; Endres and Finus 2002). Recent literature (in the context of global climate change, for example) discusses these approaches in combination, recommending a quantity ceiling as a long-term stabilisation target of greenhouse gas emissions, but then using taxes, permit trading or in some circumstances regulation to meet short-term policies, on condition that they are consistent with the long-term goals (Pizer 2002; Hepburn 2006; Stern 2007). Further developments in environmental policy analysis include a broadening of the areas of application of economic instruments beyond emissions policies and point sources (e.g., Sterner 2003), and some authors discuss economic instruments in a political economy framework (e.g., Michaelis 1996). Substantial progress has also been made in the context of life-cycle management, material flow analysis (e.g., Opschoor 1994), and the sustainability concept (e.g., Panayotou 1994; van den Bergh and van der Straaten 1994).

Nevertheless, much of the discussion in environmental economics and, specifically, economic analysis of policy instruments is still focused on environmental pollution and degradation and on the internalisation of negative externalities. Hence there is a need for more research on conservation economics compared to the extensive literature available on environmental economics and related analysis of policy instruments, a need addressed by this thesis. But what has been the prevailing focus of conservation economics so far?

In the early 1970s and 1980s, resource conservation economics concentrated on economic valuation of the benefits of preserving valuable landscapes; this was frequently used to support decision making in large public infrastructure projects based on cost-benefit-analysis

(Fisher 1981; Hampicke 1991; EPA-SAB 2009). One famous case was the Hells Canyon project in the United States, involving the construction of a hydropower dam in a wild canyon, where the benefits of preserving the landscape of the canyon were compared to the costs of alternative options for generating energy (Krutilla and Fisher 1975; Fisher 1981). When Hampicke (1991) published the first German textbook on nature conservation economics, there was much less economic analysis available that focused on nature and biodiversity conservation as such. Most of the scientific literature and empirical studies conducted in conservation economics to date have dealt with the economic valuation of the benefits of conservation. Compared to the valuation literature available, far less work exists on policy instruments in nature and biodiversity conservation. Thus a more specific need within conservation economics can be said to be policy analysis and economic instruments in conservation policies, themes which are addressed in the following chapters.

In contrast to environmental policies dealing with pollution and negative externalities, biodiversity and nature conservation policies are much more associated with the provision of public goods and services, which involves positive externalities. There are few incentives – be it for public or private actors – to engage in conservation activities when conservation costs are borne locally, whereas most conservation benefits cross local boundaries (Perrings and Gadgil 2003). If such spatial externalities or spillovers are not adequately compensated for, they may lead to an underprovision of the public goods and services concerned, such as water protection zones or nature reserves and their management (Bergmann 1999; Ring 2004).

Public support programmes, subsidies and payments for environmental services are important instruments used to compensate private actors for their conservation-related costs and the benefits they provide to society. Furthermore, taxation (e.g. through land tax exemption), charges and fees may be used to internalise these positive externalities (TEEB 2009). With regard to public actors, intergovernmental fiscal transfers constitute a suitable means to address spillover benefits in the provision of public goods and services. This category of instrument, its role in financing local services and, more specifically, the problem of spatial externalities associated with the provision of public goods and services has been extensively covered in public finance literature (e.g., Dahlby 1996; Bird 2000; Bird and Smart 2002; Bird and Tarasov 2004; Spahn 2004; Boadway and Shah 2007). However, the public finance literature does not touch (or does so very rarely), let alone cover, the field of nature conservation. This leads us to a basic research gap consisting in a lack of environmental economic, public finance and conservation research focusing on the internalisation of positive externalities in the context of conservation policies by way of intergovernmental fiscal transfers. Chapters 2 to 5 and chapters 7 and 8 contribute towards filling this gap.

Although the basic argument of the following sections and chapters builds on the economic theory of externalities, this is not to say that there is any optimal level of pollution (in the case of negative externalities), or any optimal level of conservation (in the case of positive externalities). On the one hand, environmental and conservation externalities are complex and arise in so many contexts that it is practically impossible to determine precisely their correct size. Externalities are in fact "the result of interactions between human activities and the integrated physical and biological processes of the environment" (Vatn and Bromley 1997, p. 137). The environmental pricing and standards approach, as a second-best approach in environmental economics, early on reflected and acknowledged the difficulties in determining "correct" benefits and costs in the context of environmental problems (Baumol and Oates 1971). On the other hand, given the ignorance and uncertainties attending (global) environmental change (Ring 1997), the many unknown thresholds of ecological systems, and not least continual changes in the individual and collective values of many amenities in an evolving world (Vatn and Bromley 1994), adaptive learning processes are essential, calling for policies that point in the right direction, e.g., through gradually increasing price signals in the context of ecological fiscal reform (Ring 1997), rather than assuming "once and for all correct policies" (van den Bergh and Gowdy 2000).

Building on the research needs identified above, the following two subsections explore these issues further, and highlight recent academic as well as policy developments. First, subsection 1.1.2 concentrates on fiscal instruments in general and introduces the concept of ecological fiscal reform. It is argued that relevant instruments and concepts also need to address conservation policies, which is actually a rather recent development, introduced and called for not least by the author of this thesis. Subsection 1.1.3 then moves on to fiscal instruments for biodiversity and nature conservation policies, with a focus on those instruments capable of addressing positive externalities, i.e. rewarding the benefits of conservation. The first section here covers public compensation payments geared towards private actors, while a second, more detailed section on intergovernmental fiscal transfers responds to a number of research gaps addressed by the different chapters of this thesis. Finally, subsection 1.1.4 discusses instruments for conservation policies in the wider framework of integrative biodiversity research and applied biodiversity governance.

1.1.2 From environmental taxes to ecological fiscal reform

Due to the nature of many ecological goods and services as public goods, the state – and thus public decision makers – has an essential role to play in environmental and conservation policy, including the integration of biodiversity policy into all relevant policy sectors. For public decision makers, an important means of promoting biodiversity conservation is via public finance policy and related fiscal instruments. Public finance can be defined as the way in which the government exerts influence on the economy through its public revenues and spending decisions (Musgrave et al. 1990; Zimmermann and Henke 1994; Andel 1998).

On the public revenue side, key governmental levies (*Abgaben*) include taxes, charges and fees (e.g., Hansjürgens 1992; Zimmermann and Henke 1994; Cansier 1996; Gawel 1999). Taxes are payments into the general government budget with no specific return to whoever pays the tax. They primarily serve fiscal functions. Fees and user charges are payments for specific services provided by public institutions to the individuals or firms paying the fee, such as entrance fees or fees for waste disposal. Emission or effluent charges often involve earmarking the revenues gained for environmental purposes (OECD 1997a). In Germany, the public revenues derived from so-called special charges (*Sonderabgaben*, e.g., the sewage charge) need to be earmarked for the benefit of the "group of payers" of the respective charge (Hansjürgens 1993; Cansier 1996).

Environmental levies are especially relevant for making those pay who pollute or overuse the environment and thereby contribute to the loss of biodiversity and ecosystems. However, they may also be relevant for rewarding those who conserve and sustainably use the environment when positive externalities exist. Fiscal revenues provide a number of options for reducing the existing tax burden of those who engage in biodiversity conservation, for example tax exemptions on property taxes, when a landowner agrees to relinquish his rights to future land development (Boyd et al. 2000).

On the public expenditure side, there are multiple ways to include the conservation and sustainable use of biodiversity and ecosystems more effectively in public decision making. Here, we can basically distinguish between fiscal instruments addressing public actors and those addressing private actors (Ring 2004). Public actors, i.e. local, regional and national governments, usually have tax revenues from their own sources but, depending on the country's financial constitution, may also rely on intergovernmental fiscal transfers to fulfil their public functions, such as providing schools and education, health and social care, etc. To a certain extent, the indicators used for the redistribution of tax revenues among different levels of government do indeed provide incentives to realign the behaviour of public actors. For private actors, public spending includes a wide array of subsidies, transfers and payments to individuals and businesses for different purposes (social transfers, technological subsidies, etc.).

During the last three decades there have been numerous publications and initiatives to improve the design and implementation of taxes and charges to better account for environmental pollution (e.g., Nutzinger and Zahrnt 1989; Hansjürgens 1992; Ring 1994; Ring 1997; OECD 1997a; Gawel 1999; COM 2007; EEA 2010[2]). Taking a combined look at the public revenue and expenditure side, the concept of environmental or green tax reform has attracted increasing attention (UPI 1988; Nutzinger and Zahrnt 1989; Görres et al. 1994; OECD 1997b, Schlegelmilch 1999). The basic idea of an environmental tax reform consists in introducing environmental taxes, thus greening public income, and using these additional revenues both for ecologically and socially motivated goals. Developed during the late 1980s and implemented in several European countries from the late 1990s onwards, the early focus on the public revenue side was on energy taxes (Ring 1997; BMU 2004; Dresner et al. 2006; Klok et al. 2006). This focus was soon enlarged to include other eco-taxes and fiscal policies more broadly (see Box 1.1; Ewringmann 1995; OECD 2005; World Bank 2005).

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[2] The European Environment Agency (EEA) and the OECD have jointly created an online database on economic instruments in environmental policy. The database contains information on the use of economic instruments such as environmentally related taxes and charges, environmentally motivated subsidies, tradable permits, and deposit refund systems. In addition, it outlines voluntary policy approaches such as environmental agreements, negotiated with industry, and public programmes in which firms can volunteer to participate. The database is updated at regular intervals; generally speaking, the information is more complete for taxes.

Box 1.1: Ecological fiscal reform

"… is a strategy that redirects a government's taxation and expenditure programmes to create an integrated set of incentives to support the shift to sustainable development"
National Round Table on the Environment and the Economy, Canada (2002)

Ecological fiscal reform refers to a range of taxation and pricing measures which can raise fiscal revenues while furthering environmental goals. The basic idea consists in setting incentives for resource use while providing financial resources for both ecologically and socially motivated goals. There has been much discussion about the so-called double dividend for improved environmental protection and lower unemployment (or less distortion of the tax system), i.e. using revenue from eco-taxes to reduce unemployment and labour taxes (e.g., Görres et al. 1994; Goulder 1995; Bovenberg and van der Ploeg 1998; Patuelli et al. 2005). Many organisations are now promoting the concept internationally to address environmental objectives, while creating revenues for poverty reduction and pursuing the Millennium Development Goals (e.g., OECD 2005; World Bank 2005). Only recently, the concept of ecological fiscal reform has expanded to address land-use issues, biodiversity conservation and ecosystem services provision (Meyer and Schweppe-Kraft 2006; Ring 2008a).

In the economic literature, theoretical debates around the "double dividend" of an environmental tax reform attracted much attention (e.g., Goulder 1995; Bovenberg and van der Ploeg 1998). The concept of double dividend implies the joint occurrence of an environmental and an economic benefit. Higher pollution taxes, such as energy or carbon taxes, improve the environment, while distortionary labour-related taxes can be lowered through revenue-recycling, eventually boosting employment (see Bosquet 2000 and Schöb 2005 for overviews). Patuelli et al. (2005) present a meta-analytical synthesis of simulation studies on environmental tax reform and the double dividend. They conclude "that the total effect of an environmental tax-and-recycle policy has a significant influence on the economic variables (second dividend) when employment is used" (ibid., p. 564) but is much less clear, when effects on GDP are analysed.

It comes as no surprise that the exact definition of the "double dividend" and the design of the simulation study do influence the results. Therefore, Patuelli et al. (2005) strongly recommend that policy design and implementation should not be based on just one type of economic model to simulate the effects of an environmental tax reform. Similarly, van den Bergh and Gowdy (2000) reason that the potential double dividend may be systematically underestimated with equilibrium analysis, recommending evolutionary models to shed light on long-term processes such as large-scale tax revisions over long periods.

Taking an ecological or biodiversity-related view of fiscal reform is relatively recent and needs to be strengthened in order to account for the positive externalities of conservation policies (Meyer and Schweppe-Kraft 2006; Ring 2008a). Taking this broader view of ecological fiscal reform, fiscal instruments are increasingly being used as effective mechanisms to achieve conservation objectives, to provide incentives for conservation, and to raise funds for conservation purposes (OECD 1999; Emerton et al. 2006). Beyond that, fiscal instruments are central to social policies and the redistribution of wealth and income. Thus they are especially suited to combine biodiversity and ecosystem conservation with sustainable livelihood issues and poverty reduction (e.g., OECD 2005; World Bank 2005), an indispensable perspective for the design and implementation of policy instruments in developing countries.

1.1.3 Fiscal instruments addressing positive externalities in conservation policies

Due to the non-linearities, complementarities, uncertainties and often unknown thresholds associated with ecological systems (MA 2005a; TEEB 2008; EPA-SAB 2009; Naeem et al. 2009; Ring et al. 2010b), the ecological foundations of conservation policies are more complex compared to many other environmental issues. This has consequences for economic analysis, and the economic foundations of conservation policies are hampered by a number of considerations.

Economic valuation as one basis for conservation policies may be limited by ecological, but also by social and cultural considerations (Turner et al. 2003; Ring et al. 2010b). Economic valuation builds on marginal analysis and thus involves the substitutability of the object to be valued. If species or ecological processes are complementary to each other and are not substitutable, a key requirement for marginal economic analysis is lacking from an ecological perspective (Ring 1997; Balmford et al. 2008). From a social or collective choice perspective, social norms and institutions are crucial for societal decision making (Vatn and Bromley 1994). For example, both societies and citizens may decide to put an infinite value on certain ecological goods for cultural or religious reasons, be it a species, a sacred wood or a special cultural landscape, and not expose it to a cost-benefit calculus (e.g., Spash 2000). In both cases, decision making needs to consider multiple criteria, building on ecological, economic and social considerations. Furthermore, non-use values, such as existence, bequest and, to some extent, option values play a much greater role in conservation policies compared to other environmental issues. All these characteristics make it more difficult to define conservation issues through market mechanisms alone (Hampicke 1991; Gowdy 1997; OECD 1999; Turner et al. 2003; MA 2005b; Spash and Vatn 2006; TEEB 2008).

Nevertheless, as in environmental policy, many goods and services associated with nature and biodiversity conservation and their sustainable use are public goods and services. Thus governments from the local up to the global level play a powerful role in valuing nature's "public" services to society as a whole, by issuing regulation and property rights, through economic instruments as well as informational and communicative instruments, to name but a few (e.g., McNeely 1988; OECD 1999; Sterner 2003; Unnerstall 2004; MA 2005b; TEEB 2009; Ring et al. 2010a).

Over many decades, continuous improvements have been undertaken to realign perverse subsidies and to use taxes, charges and tradable permits to internalise negative externalities (EEA 2010). The complementary task of internalising the positive externalities of providing ecological public goods and services has gained comparatively less attention[3], although greater moves in this direction have become apparent more recently (OECD 1999; MA 2005b; TEEB 2008). Taxes, charges and fees as well as intergovernmental fiscal transfers, subsidies, public support programmes and compensation payments are important instruments to reward the often unrecognised benefits of biodiversity conservation (Emerton et al. 2006; TEEB 2009). During the last decade – and elaborated first in a developing countries context – the concept of "Payments for Environmental Services" (PES) finally put the internalisation of positive externalities centre stage (Landell-Mills and Porras 2002; Wunder et al. 2008).

It is against this background that I will discuss in more detail two types of fiscal instruments involving public expenditure: government-financed compensation payments to private actors and intergovernmental fiscal transfers addressing public actors.

1.1.3.1 Compensating land users for environmental services

In the European Union (EU) and its member states, as well as in many other countries, public compensation payments for the provision of environmental goods and services focus almost exclusively on private land users, be it in agriculture, forestry or aquaculture (Ring 2008b). Agri-environmental and conservation programmes need to be continuously monitored and improved to increase their ecological effectiveness and economic efficiency (Baldock et al. 2002; Wätzold and Schwerdtner 2005; Hampicke et al. 2006; Claasen et al. 2008; Wätzold et al. 2008b). In contrast to the bulk of intergovernmental fiscal transfers, which are provided as lump-sum transfers, such compensation schemes involve specific payments to land users that are to be spent for a given purpose. The European agri-environmental and conservation schemes usually require matching funds on the side of the implementing national and regional governments, thereby increasing the overall budget of the respective public programme. Through these programmes, land users can apply for compensation payments that compensate them for the benefits they forego through using less intensive agricultural practices or through implementing active conservation measures.

One crucial task of all these programmes is the identification of a baseline for payments: What measures can be expected from land users as standard, apart from any claims for compensation? The continuing debate on the "principles of good agricultural practice" (*gute fachliche Praxis der Landwirtschaft*) and related minimum conservation standards in German nature conservation law reflects the difficulties involved in reaching a societal consensus on these controversial issues (Bauer et al. 1996). Nevertheless, a baseline and its enforcement are essential from an economic perspective, because of its influence on the ecological effectiveness and efficiency of the respective programme. Another major problem lies in the strategic behaviour of participating land users who have an incentive to claim higher compensation costs than those actually incurred, because their compensation payment is based on (estimated) conservation costs. Regulators in turn may encounter significant transaction costs in

[3] For early exceptions from a conservation NGO background, cf. McNeely (1988).

terms of information and negotiation costs for specifically tailored compensation payments that reflect the actual conservation costs of programme participants (Wätzold and Schwerdtner 2005).

Land users may also benefit from public compensation programmes that account for damage caused by wildlife. The aim of damage compensation payments is to compensate land users (mostly farmers and fishermen) partly – or, in exceptional cases, fully – for the damage caused by protected species. Examples include wolf compensation schemes, which compensate for damage to livestock (Fourli 1999), and compensation for damage to fishing gear caused by seals (chapter 9, Similä et al. 2006). Although compensation payments for wildlife damage can be essential in preventing the culling of protected and highly endangered species, they are often associated with – or even create – a negative perception of wildlife due to their focus on "damage". In this context a change of perspective can be identified in many countries, away from damage compensation schemes towards developing public payments related to the abundance of wild animals or towards supporting measures to provide feeding habitats for protected species (chapter 9; Schwerdtner and Gruber 2007). In this way, rather than being compensated for damage caused by otters feeding on commercial carp, a carp pond farmer may be rewarded for the environmental service of "feeding" otters with extra carp. This extra carp is provided with the help of public monies, based on society's demand that the otter should be strictly protected.

However, as in the agri-environmental and conservation programmes mentioned above, strategic behaviour and windfall profits made by participating land users need to be taken into account when it comes to both damage compensation and environmental service payments. Regarding damage compensation schemes, regulators may again encounter high transaction costs due to the difficulty of assessing the actual damage caused by the protected species covered in the programme (the damage may be caused by another species or even by bad management practices). For environmental service payments, a regular monitoring of species is required in order to relate compensation to species abundance. Schwerdtner and Gruber (2007) discuss the pros and cons between these two options in terms of their related transaction costs.

At the global scale, "Payments for Environmental Services" (PES) developed as a mechanism to pay land users for the provision of environmental services, compensating them for the conservation benefits of environmentally sound land management, which implies higher local costs (Landell-Mills and Porras 2002; Wunder 2005; Pagiola and Platais 2007; Engel et al. 2008; TEEB 2009). Wunder (2005, p. 3) defines a PES scheme as "a *voluntary* transaction where a *well-defined environmental service* (or land use likely to secure that service) is being 'bought' by a (minimum one) *service buyer* from a (minimum one) *service provider* if and only if the service provider secures service provision (*conditionality*)." The conditionality of environmental service provision relates to the above-mentioned baseline of compensation payments. A service provider needs to provide an environmental service that is additional to existing behaviour or practices in order to be eligible for the payment.

The PES concept originated in the context of developing countries and was associated more with private-private interactions, involving market transactions around environmental services provided by private agents. For example, a company providing drinking water to large cities pays compensation to upstream land users for maintaining forest cover or using less fertiliser as a means of keeping drinking water as clean as possible. However, many PES schemes involve public agencies and payments from public budgets to private land users. The PES programme in Costa Rica, which is widely regarded in the literature as an early and innovative programme, provides public compensation payments to land users for environmental services such as greenhouse gas mitigation, water services, scenic value and biodiversity (Landell-Mills and Porras 2002; Pagiola 2008). In this sense, "traditional" agri-environmental programmes in Europe and the United States can also be regarded as payments for environmental services (e.g., Baylis et al. 2008; Ring 2008c; Vatn 2008).

From an economic perspective, the efficiency of PES programmes and the various criteria influencing it have been a focus of interest (Pagiola 2005; Claasen et al. 2008). Wunder et al. (2008) have analysed a set of user- and government-financed PES programmes in developed and developing countries with respect to design, costs, environmental effectiveness and livelihood outcomes. They found that user-financed PES schemes were generally better targeted and more closely adapted to the local conditions and requirements. The schemes also had better monitoring and a greater willingness to enforce conditionality. Government-financed programmes, by contrast, sought to achieve far more additional objectives than user-financed programmes, thereby lowering their effectiveness and the efficiency of environmental service provision.

1.1.3.2 Towards ecological intergovernmental fiscal transfers

In federal systems, tax revenues are redistributed from national to state and further on to local governments in order to provide decentralised jurisdictions with monies to fulfil their public functions: building schools and hospitals, or constructing and maintaining roads. These intergovernmental fiscal transfers serve allocative, distributive and fiscal functions (e.g., Lenk 1993; Zimmermann 1999; Boadway and Shah 2007). Their specific design is regulated in financial constitutions and fiscal equalisation laws. Fiscal transfer systems are located in the realm of ministries of finance, which rarely consider ecological matters (Ring 2002). The same holds for public finance literature on intergovernmental fiscal transfers, where environmental and conservation issues are of no specific interest (e.g., Bird and Tarasov 2004; Boadway and Shah 2007). Environmental, resource and ecological economic literature in turn have largely missed this type of policy instrument in its potential for realising ecological objectives. Consequently, ecological fiscal transfers addressing long-term and precautionary conservation issues between different levels of government rarely exist in environmental and conservation policies or real-world politics, an insight resulting from analysis in chapters 2 and 3 (Ring 2001, 2002).

For example, Sterner (2003) in his otherwise very comprehensive book on *Policy Instruments for Environmental and Natural Resource Management* does not even include the term fiscal transfer or fiscal grants in his index. Public budget-financed payments address only private agents (ibid., p. 102ff.). Although he covers the direct provision of public goods by public agencies (ibid., p. 74f.) and acknowledges, among other things, the provision and maintenance of natural parks, only the traditional instruments of taxes, charges and fees are mentioned as ways of internalising negative externalities. Internationally, little scholarly literature exists on the use of intergovernmental fiscal transfers to achieve conservation objectives. Where such transfers are supported theoretically, few options have been suggested regarding their design and implementation. Thus intergovernmental fiscal transfers represent an innovative instrument in conservation policies in particular, requiring advances in both environmental and ecological economic theory as well as applied research.

In Germany, some initial steps have been taken towards promoting ecological fiscal transfers. The expert council for the environment has been recommending the use of intergovernmental fiscal transfers to acknowledge the provision of environmental goods and services especially by rural and remote areas since the mid-1990s (SRU 1996; Ewers et al. 1997). The Federal Agency for Nature Conservation has principally been interested in assessing the potential of intergovernmental fiscal transfers for conservation objectives, evidenced so far in its request for a brief expert comment on federal-state relationships (Seitz 2001) and in an initial study it commissioned on state-local relationships (Perner and Thöne 2005; Perner and Thöne 2007). Seitz, in his comment from a public finance view, does not acknowledge any potential for considering conservation issues in German fiscal equalisation between the federal and the state governmental level (*Länderfinanzausgleich*).[4] By contrast, legal scholars Czybulka and Luttmann (2005), drawing on their expert statement for the German Green Party, and Kloepfer (2006) argue in favour of integrating conservation into German fiscal equalisation from the federal to the state level. From a public finances and planning perspective, Perner and Thöne (2005, 2007) suggest the very first options for integrating nature conservation into fiscal equalisation at the local level in Germany. However, none of the few existing proposals have been considered for integration into the German fiscal transfer system so far. Alternative options and further arguments substantiating the integration of nature conservation into fiscal transfer systems are required, including the development of suitable conservation indicators.

In this habilitation thesis, the need for integrating nature conservation issues into intergovernmental fiscal transfers is substantiated from an economic and ecological perspective. Building on the economic theory of federalism and on insights from sustainability science, starting points for such integration are presented in chapters 2 (Ring 2002) and 3 (Ring 2001). For this purpose, the thesis also provides – for the first time – an empirical review in chapters 2 and 3, updated by chapter 4 (Ring 2008a), of how and to what extent ecological public func-

[4] A similar tendency has been observed for environmental taxes and the concept of ecological fiscal reform in the late 1980s and early 1990s. Once certain environmental policies become political reality – usually pushed by stakeholder groups outside the realm of public finance – options for such instruments tend to be discussed more optimistically, even within public finance. However, this is a hypothesis deriving from more than 20 years of research in this field; providing scholarly evidence for it through a careful analysis of public finance literature in combination with the actual design and implementation of policies would require yet another study.

tions have thus far been integrated into German fiscal equalisation; this is done by means of a textual analysis of the 13 fiscal equalisation laws in force in the German *Länder*.

An important issue in designing intergovernmental fiscal transfers relates to the choice of indicators for their distribution. This task holds for any public goods and services alike. For specific transfers, the selection of indicators is quite straightforward. They are usually based on objective indicators closely related to the purpose of the specific transfer. The more substantive part of intergovernmental fiscal transfers is distributed as general or lump-sum transfers, on the basis of the fiscal capacity and the fiscal needs of the respective jurisdictions. Whereas fiscal capacity is comparatively easy to determine, the adequate fiscal need of a jurisdiction has been the focus of much research and political debate from both a theoretical and practical perspective (Zimmermann 1996; Shah 2007a). The choice and quantification of suitable indicators, as well as their adaptability to changing economic, political and social conditions keep coming up as relevant research questions. A basic tension exists between the direct identification of a more realistic and "adequate" fiscal need of a jurisdiction on the one hand and approaches that build on widely available indicators such as inhabitants or area of a jurisdiction on the other, both of the latter being commonly used indicators for identifying fiscal needs in many countries. Although the "direct" identification of fiscal needs may be more accurate, it nonetheless entails systematic weaknesses and political disadvantages (Lenk 2009, p. 29). The indicator-based identification of fiscal needs, although a second-best solution, continues to be the more transparent and flexible system.

The few existing studies on the integration of conservation-related public functions into intergovernmental fiscal transfers propose comparatively complicated indicators. Köllner et al. (2002) present a biodiversity-related indicator for amending the fiscal transfer system in Switzerland (a rather difficult approach per se due to the complexity and manifold biodiversity indicators available), whereas Perner and Thöne (2007) suggest a so-called "eco-points" approach, based on landscape planning procedures and relevant indicators. However, landscape plans – an additional feature in German local and regional land-use planning addressing nature conservation and landscape protection – are not even available for all municipalities in Germany.

Thus building on the empirical review of ecological public functions in German fiscal equalisation laws, this thesis develops and applies an easy and widely available area-related indicator for integrating nature conservation into intergovernmental fiscal transfers. Taking as an example the fiscal transfer system at the local level in Saxony, a federal state (*Land*) in Germany, the fiscal impacts for Saxon municipalities of using this indicator are modelled in a spatially explicit way for the fiscal year 2002 in chapter 4 (Ring 2008a), presenting two different options for using this indicator.

Spatially explicit modelling and GIS tools can help to illustrate the consequences of ecological fiscal transfers where they have not yet been introduced. Fiscal transfer schemes are country-specific and highly politicised due to the substantial financial flows involved. Building on existing fiscal transfer schemes and integrating suitable ecological indicators can help decision makers in promoting innovative solutions. For Switzerland, Köllner et al. (2002) developed a model based on biodiversity indicators and cantonal benchmarking for intergovernmental fiscal transfers. Ring (2008a) suggested ways of incorporating protected areas into the intergovernmental fiscal transfer system at the local level in Saxony, Germany. Such a spatially explicit analysis of the fiscal impacts of ecological fiscal transfers for local governments, based on a real-world fiscal transfer system, has not been published previously.

For the development of a suitable indicator, it also proved helpful to take a look outside Europe and investigate fiscal transfer systems in Latin America. In Brazil, several states have already introduced "conservation units", a protected area-based indicator, for the redistribution of value-added tax (ICMS) from state level to municipalities, starting from 1992 (Ring 2008b). So far, little to no notice has been taken in Germany and other European countries of experiences with the ecological ICMS in Brazil. Most of the early literature on this new instrument is written in Portuguese (e.g., Loureiro 1998; 2002) and has been made available to an international readership only through grey literature and one book chapter in English (Grieg-Gran 2000; May et al. 2002). Chapter 5 of this thesis presents a thorough review of the ecological ICMS, including an examination of the design and implementation of this oldest scheme of ecological fiscal transfers to the local level in terms of its fiscal and ecological effects (Ring 2008b). In addition, the potential of the scheme to be transferred to other federal countries is examined, based on the experience of those Brazilian states that have implemented the ecological ICMS so far.

Portugal has very recently become the first European country to amend its local government financing law to account for Natura 2000 sites, the European Union's network for protected areas of major community significance (Prates 2008; Ring 2008a; Ferreira 2009).[5] Thus both Brazil and Portugal use protected areas as an easily available indicator to account for biodiversity and nature conservation. Many countries, including several German states, already use the area of a municipality, district or province as an indicator for assigning fiscal transfers (Ring 2002, 2008a), so it is only a small and cost-effective step to consider "protected area" as a basis for an indicator that takes account of nature conservation in fiscal transfer mechanisms. In contrast to the landscape planning indicators proposed by Perner and Thöne (2005), an indicator based on protected areas is already available statistically and can be directly introduced into existing fiscal transfer procedures.

Indicators for the redistribution of tax revenues among different levels of government also provide incentives to realign the behaviour of public actors, as the increase of protected areas in Brazilian states with ecological fiscal transfers has shown (see chapter 5). However, in most countries today, this kind of incentive mechanism works only to attract more businesses, inhabitants and construction projects, followed by land-use activities that destroy, fragment and degrade natural habitats and thus serve to exacerbate the continuing loss of biodiversity. The management and opportunity costs associated with protected areas powerfully shape their

[5] Given the Brazilian experience, it may be no coincidence that a Portuguese speaking country in Europe took this step first.

image as an obstacle to economic development, leading to commonly encountered local opposition to protected areas (Bauer et al. 1996; Stoll-Kleemann 2001). If intergovernmental fiscal transfers included consideration of biodiversity and ecosystem-related indicators when allocating taxes to lower governmental levels, public decision makers might be expected to take care of nature as a basis for public revenues in a similar way that they take care of their other tax bases today.

Choosing the right indicators is but one task associated with fiscal instruments; the other relates to their concrete design and implementation. In this context, governance research plays an increasingly important role: proposals to solve biodiversity problems are of little use, if they neglect to take account of governance issues. This leads us to the next topic of biodiversity governance.

1.1.4 Biodiversity governance: Coordinating institutions, levels and actors for the conservation and sustainable use of nature and biodiversity

At the core of governance research lies the desire to understand how decisions are made and what furthers their successful implementation (Kissling-Näf and Kern 2002). Notwithstanding the manifold uses of the governance concept in recent years, the two major approaches to governance research relate to the political and social sciences on the one hand and to economics on the other (Brunnengräber et al. 2004). The understanding of governance in economic literature has been strongly influenced by Coase (1937; 1960) and by Williamson (e.g., 1979; 1996) who, with their work on transaction cost economics and the governance of contractual relations, contributed greatly towards shaping the field of New Institutional Economics. Ostrom, a political scientist, strongly shaped economic governance research with her theoretical and empirical work on the evolution of institutions for collective action, which she applied to the management of common pool resources such as forests, fisheries, grazing lands, and irrigation systems (see Ostrom 1990; 1999; 2005). She has been regarded as representing a separate school of public choice theory (Mitchell 1988) and is a good example of the close interrelationship between political science and economics in governance research. Another strand of economic governance research can be found in political economy, also a field closely related to political science. According to Lütz (2004), the discourse on governance in political economy centres on the characteristics and functioning of modern market economies and their governing institutions.

Although fiscal federalism deals explicitly with the economic analysis of decision making at different levels of government, it has been much less discussed in the context of governance research. As a subfield of public finance, fiscal federalism is concerned with understanding "which functions and instruments are best centralised and which are best placed in the sphere of decentralised levels of government" (Oates 1999b, p. 1120). In other words, fiscal federalism deals with the assignment of public functions, expenditures and revenues to the proper level of government (Musgrave 1959; Oates 1972). Thus the design and implementation of intergovernmental fiscal relations, including the incentive effects of fiscal instruments on the jurisdictions, has received considerable attention in the field of public finance. However, the institutional arrangements surrounding policy making and administration, which vary widely in terms of the form and membership of relevant decision-making bodies, have not yet received adequate attention. In addition to the incentive regime associated with governance structures, the success of different institutional arrangements also depends on the interactions of those structures with other formal and informal institutions in a country (Shah 2007b).

Some work has been done, under the heading of "environmental federalism", on applying the economic theory of federalism, its principles and approaches, to environmental problems (e.g., Zimmermann and Kahlenborn 1994; Oates 1999b; Oates 2001; Kloepfer 2002). However, few authors explicitly link environmental federalism to the concept of governance, a gap that is addressed in chapter 8 of this habilitation thesis (Ring 2008c). Given the challenge of managing globalisation and providing global public goods, the international literature has only recently begun to take a more comprehensive approach to public finance (Kaul et al. 2003). In response to global challenges such as climate change and the global loss of biodiversity – to name but the most pressing environmental challenges on the global scale – a New Public Finance has emerged that addresses economic, social *and* environmental problems, explicitly taking account of *sustainability* issues (Kaul and Conceição 2006a). The new approach to public finance explicitly encompasses a governance perspective, addressing local, national and global issues, as well as public and private sector interaction relating to the provision of goods and services. Applying this perspective to nature and biodiversity conservation constitutes one of the major innovative contributions of this thesis.

Environmental governance is defined by Paavola and Adger (2005) as the management of all environmental resources, including renewable and non-renewable resources such as minerals, soils, water and forests. It also includes environmental resources that have been the focus of more recent interest, such as biodiversity, the ozone layer and atmospheric sinks. Environmental governance involves multi-level and multi-actor governance, i.e. the coordination of environmental decisions and various actors across governmental levels, from the local, regional and national levels up to the international level (Ring 2008c). During the last decade the concept of governance has attracted substantial attention from the environmental sciences, not least as a concept that facilitates an integrative approach to sustainable development and, more specifically, sustainable land use from the perspectives of various social scientific disciplines (Adger et al. 2003; Brunnengräber et al. 2004; Köck 2005; Görg 2007).

"Biodiversity governance" is understood here as a subfield of environmental governance, including the analysis of the formal and informal institutions of nature and biodiversity conservation as well as the related interplay of multiple governmental levels and actors. The design and implementation of policy instruments for their conservation and sustainable use represent a challenge in federal systems, where various public and private actors at different levels of government are engaged in continual interaction (Ring 2008c). This applies even more in times of global environmental change, with climate change and biodiversity loss calling for international collective action and subsequent implementation of relevant policy goals and instruments at European, national, subnational and local levels (MA 2005a; Stern 2007; TEEB 2009).

Literature on linking nature and biodiversity conservation to the economic theory of federalism is comparatively rare, a gap that is addressed in the following chapters. Within this context, the spatial distribution of conservation benefits and costs deserves special attention (Döring 1998; Revesz 2000; List et al. 2002; Wätzold et al. 2008a). The local costs versus global benefits of many conservation-related public goods and services are a key justification for designing policy instruments that internalise these spillover benefits in order to ensure adequate provision of these goods and services (Perrings and Gadgil 2003; see chapters 7 and 8; Ring 2004, 2008c). Regarding the choice of instrument, intergovernmental fiscal transfers to local governments are a suitable means of internalising these spatial externalities in the public sector; for private actors, compensation payments or payments for environmental services can offer incentives for providing the services required.

Looking at policies and instruments through a governance lens reveals new perspectives for economic analysis in the field of public finance (cf. Kaul and Conceição 2006b):

Firstly, the analysis of governance structures focuses on the division of responsibilities between governmental levels. The assignment of public functions to the appropriate levels of government is a familiar topic in public finances, and fiscal instruments such as intergovernmental fiscal transfers are specifically designed to address mismatches between functions and the financial resources available to fulfil these functions. The perspective missing in this respect is a more systematic treatment of ecological public functions and their integration into intergovernmental fiscal transfers (see chapter 2; Ring 2002). With respect to compensation payments in conservation policies, the coordination of instruments between governmental levels is far from a trivial task. For example, as demonstrated in chapter 9 (Similä et al. 2006), European framework regulation relating to conservation policy is highly complex in terms of its interplay with European state aid regulation, the purpose of which is to avoid market distortion. European Union member states tend to interpret conservation and state aid regulations in different ways, leading to damage compensation programmes in some countries losing EU approval due to the state aid regulations in force. Related to the issue of multi-level governance is the problem of potential mismatches between policy instruments and the ecological scale of the conservation challenges the instruments aim to address (Levin 2000; Young 2002; Cash et al. 2006).

Secondly, governance structures are analysed with respect to responsibilities divided between public institutions, civil society and the private sector (see chapter 6 and 8; Jentsch et al. 2003; Ring 2008c). Conservation policies and instruments are, by their very nature, a result of hierarchical decision making, because they are issued by the public sector. However, the use of economic instruments in conservation policies already introduces a market-oriented element. Furthermore, any policy instrument will ultimately prompt reaction by its addressees and other indivuals and groups affected (or even non-affected), be they local governments, non-governmental organisations, companies, land users or citizens. Policy analysis in a governance perspective is thus not restricted to the instrument in question, its primary addressees and intended objectives. Instead, biodiversity governance assumes a more comprehensive perspective and includes the interplay between the public and private sector, and between different actors. For example, in the case of ecological fiscal transfers in Brazil, policy analysis would always address fiscal impacts on local governments and ecological effects relating to environmental and conservation objectives. The governance perspective offers a more explicit treatment of relationships between governmental levels and addresses the interplay between different societal actors – in this case, the evolving public-private partnerships between local governments and land user associations that arise from the introduction of ecological fiscal transfers (see chapter 8; Ring 2008c).

These thoughts on biodiversity governance bring to a close the first section on economic instruments in environmental and conservation policies. The following two sections of chapter 1 present the background to and main conclusions from the published articles constituting this habilitation thesis. The articles are grouped into two main sections (sections 1.2 and 1.3), mirroring the structure of the thesis.

1.2 Integrating local ecological services into intergovernmental fiscal transfers

This section refers to Part II of the thesis, encompassing chapters 2–5. It addresses a selection of articles on the design and use of intergovernmental fiscal transfers in environmental and conservation policies, breaking new ground at the interface of environmental economics and public finance.

Sustainable development is closely linked to land-use practices. Sustainable land use in turn requires a variety of conservation efforts and services, not least at the local level. Despite this, there are few incentives for local actors to engage in environmental and conservation activities when costs are borne predominantly at the local level, whereas ecological benefits cross local boundaries (Perrings and Gadgil 2003). This is the case for a number of ecological services, for example water protection or nature reserves. Decisions on the designation of the respective protected areas are often taken by institutions above the local level, whereas the practical, everyday consequences in terms of management measures or land-use restrictions are borne by local actors, often without any – or with insufficient – compensation. Such spatial externalities or spillover benefits – if not adequately compensated – lead to an underprovision of the public goods and services concerned (Bergmann 1999; Ring 2004).

In contrast to the vast literature available on compensation payments for land users, comparatively few publications exist with regard to local public actors. However, it is just as important to consider local governments, because they hold decision-making, financing and implementing competencies for a number of land use-related issues. Although these competencies differ from country to country, land-use planning competencies in particular along with the implementation of various land-related measures are often decentralised and entail considerable consequences for conservation outcomes.

In the subsections to follow, each of the four articles in Part II ("Integrating local ecological services into intergovernmental fiscal transfers") is briefly introduced and the main results presented. Chapters 2 and 3 substantiate the need for integrating nature conservation issues into intergovernmental fiscal transfers from an economic and ecological perspective. In addition, starting points for such integration are presented, building on the economic theory of federalism and insights from sustainability science. In chapter 4, suitable indicators are devised for integrating nature conservation into intergovernmental fiscal transfers. By way of an example, the fiscal impacts of using these indicators are modelled in a spatially explicit way for fiscal equalisation at the local level in Saxony, a federal state (*Land*) in Germany. Finally, chapter 5 investigates the design and implementation of existing ecological fiscal transfers to the local level in Brazil, along with their fiscal and ecological effects, and examines the potential to transfer this system to other federal countries.

Meanwhile, national and international studies increasingly draw on the material provided by chapters 2, 4 and 5 of this thesis (e.g., Swedish Environmental Protection Agency 2005; Hajkowicz 2007; Fernandes 2008; Ferreira 2009; Irawan and Tacconi 2009; Kamal-Chaoui and Robert 2009; Köllner 2009; Kumar and Managi 2009a, b; TEEB 2009).

1.2.1 Ecological public functions and fiscal equalisation at the local level in Germany

This chapter was published as "Ring, I. (2002). Ecological public functions and fiscal equalisation at the local level in Germany. Ecological Economics 42(3): 415–427" (Impact factor 2008: 1,912).

The article starts from the assumption that sustainable development requires the acknowledgement and appropriate financing of ecological, economic and social public functions at all levels of government. Whereas the significance of socio-economic functions has a comparably long tradition in federal systems, a corresponding consideration of ecological functions has yet to be fully realised. In this context, the article introduces and defines the term "ecological public functions", a concept new to the literature at this point. Building on the definition provided by Ring (2002), it is now referred to and used in international scientific literature (Holmgren 2006; Hajkowicz 2007; Fernandes 2008; Kumar and Managi 2009a, 2009b).

Continuing on a conceptual level, the article presents essential public finance principles such as the decentralisation rule for the provision of public goods and services (Musgrave 1959; Oates 1972) and the principle of fiscal equivalence (Olson 1969) and links them to the provision of ecological public goods and services. Starting with a concise review of environmental federalism (based, e.g., on Huckestein 1993; Hansjürgens 1996; Scheberle 1997; Oates 1999b), the spatial dimension of environmental resources (Smith et al. 1997) and the mobility of environmental compartments (e.g., Hansjürgens 1996 for highly mobile air pollutants) receive special attention in the process of selecting appropriate levels of government for the provision of ecological goods and services. Although public goods and services associated with less mobile environmental compartments – e.g., many land use-related services, such as water resources protection and nature conservation – are more efficiently provided at decentralised levels, spatial externalities often exist which require appropriate solutions. Given that the diseconomies of large-scale operation call for local provision of ecological services, these externalities may be internalised through intergovernmental fiscal transfers, compensating local governments for the spillover benefits of its local expenditure. As already suggested by Olson (1969) for other public goods (such as education), this solution can readily be applied to ecological public goods and services.

Building on these conceptual foundations, the article presents original empirical research. The status quo of and perspectives for integrating ecological public functions into intergovernmental fiscal transfers to the local level are investigated. For this purpose, the 13 intergovernmental fiscal transfer laws of the German *Länder* are analysed with regard to their consideration of environment- or conservation-related public functions (or their lack of the same). Area-related indicators are identified as an initial, indirect approach to acknowledging ecological public functions. However, by far the most widely used option for considering ecological public goods and services are special purpose matching grants. The article concludes with a number of policy options aimed at taking better account of ecological public functions in German fiscal equalisation at the local level.

1.2.2 Ökologische Aufgaben und ihre Berücksichtigung im kommunalen Finanzausgleich

This chapter was published as "Ring, I. (2001). Ökologische Aufgaben und ihre Berücksichtigung im kommunalen Finanzausgleich. In: Spehl, H., Held, M. (Eds.), Vom Wert der Vielfalt. Zeitschrift für angewandte Umweltforschung, Sonderheft 13: 236–249".

While Ring (2002) was written for an international readership, Ring (2001) was published in a widely read German journal for applied environmental research. The latter article is more specific regarding the presentation and examination of the German fiscal transfer system as well as the suggestions put forward for ecological fiscal transfers. Intergovernmental fiscal transfer systems are highly context-specific. Thus a German scholar or decision maker will naturally expect – and require – more detailed knowledge of the German federal system; this is of less interest to a foreign scholar, who will be more interested in insights of a generic nature and in their suitability for transfer.

The local governmental level in Germany is characterised by a comparatively high level of autonomy, with far-reaching planning authority related to communal land-use issues. Local governments have their own public revenue sources, consisting mainly of real estate and business taxes, and their local share of income taxes. Depending on a municipality's fiscal capacity, intergovernmental fiscal transfers on average account for up to 30 % of overall municipal revenues in western Germany or even 50 % in eastern Germany (Karrenberg and Münstermann 2000).

Given the importance of intergovernmental fiscal transfers for the provision of local public goods and services, ecological public functions need to be given adequate consideration. Since the 1970s, German environmental and conservation policies have been realised through a complex legal and institutional framework, which assigned numerous functions to different governmental levels. However, these policies still suffer from insufficient financial resources compared to other, more traditional policy fields. The situation is even more serious for conservation-oriented tasks compared to environmental pollution with its technical and infrastructure-oriented bias (Kaule 1991; Henle 1995; Steffens 1997). Early German publications proposed ecological fiscal transfers in the context of sustainable land use and the acknowledgement of ecological services provided by rural areas to urban agglomerations (SRU 1996; Ewers et al. 1997; Bergmann 1999; Rose 1999).

Based on a status quo analysis of the various German fiscal equalisation laws at the local level, a number of options are presented for systematically integrating neglected ecological functions and indicators into the existing legal framework. The empirical analysis shows that end-of-pipe and infrastructure-related ecological functions are already firmly integrated by way of conditional grants. However, precautionary functions such as nature and resource conservation issues are still largely absent. Therefore, a comprehensive consideration of ecological public functions in Germany calls for the integration of nature conservation issues and corresponding indicators into the system of intergovernmental fiscal relations. As regards the types of grants used, there is an obvious lack of general, lump-sum transfers based on conservation-related indicators comparable to the role played by existing socio-economic indicators.

1.2.3 Compensating municipalities for protected areas. Fiscal transfers for biodiversity conservation in Saxony, Germany

This chapter was published as "Ring, I. (2008a). Compensating municipalities for protected areas. Fiscal transfers for biodiversity conservation in Saxony, Germany. GAIA 17/S1: 143–151" (Impact factor 2008: 0,653).

The years prior to this publication saw conceptual developments and more probing analyses of ecological fiscal transfers, especially in Europe. Köllner et al. (2002) developed a model based on biodiversity indicators and cantonal benchmarking for fiscal transfers in Switzerland. Perner and Thöne (2005), in a study commissioned by the German Federal Agency for Nature Conservation, suggested ways of including nature conservation in communal fiscal transfers. Finally, Portugal actually implemented conservation-related indicators in real-world policy making, the first European country to do so. With effect from January 2007, Portugal amended its communal financing law, acknowledging Natura 2000 sites and other protected areas as indicators for intergovernmental fiscal transfers from the national to the local level (de Melo and Prates 2007; Prates 2008; Ferreira 2009).

In addition to summarising these trends, Ring (2008a) presents an updated investigation of ecological fiscal transfers in Germany since 2006, using the same empirical methodology as in Ring (2001, 2002). In the meantime, all sparsely populated states in Germany (Brandenburg, Mecklenburg-Western Pomerania and Saxony-Anhalt) had begun to use land area as an indicator for general lump-sum transfers, in addition to Rhineland-Palatinate, a state with a higher average population density. Earmarked fiscal transfers still represent the most common method of including ecological public functions in communal fiscal transfers.

The main body of this article introduces a model of the intergovernmental fiscal transfer system from state level to local level in Saxony, Germany, one that has been enlarged to include designated protected areas: national and nature parks, EU-Natura 2000 sites, biosphere, nature and landscape reserves. Apart from the introduction of an area-based ecological indicator, the model reproduces intergovernmental fiscal transfers in Saxony as of 2002, building on real social and economic data. In order to obtain the ecological indicator, nature conservation units (CUs) were identified in each municipality by overlaying Geographic Information System (GIS) layers of the various categories of protected areas over a total of 537 municipal borders in Saxony.

Two ways of integrating CUs into the Saxon fiscal transfer system are presented along with the modelling results. The first model considers CUs within the calculation of general lump-sum transfers by "translating" CUs into the generic indicator of inhabitants. Thus in addition to the main existing approach for determining municipal fiscal needs based on inhabitants and an additional approach specific to Saxony based on schoolchildren, a new additional "conservation approach" is suggested. In the second model, a specified amount of the overall transfer sum is set aside for distribution according to the share of CUs a municipality holds in relation to its total area. Both models lead – albeit to varying degrees – to higher transfers to rural communities in particular, thereby acknowledging the latter's ecological services to society. Using a GIS, results for both models are presented in a spatially explicit way in the form of maps.

1.2.4 Integrating local ecological services into intergovernmental fiscal transfers: The case of the ecological ICMS in Brazil

This chapter was published as "Ring, I. (2008b). Integrating local ecological services into intergovernmental fiscal transfers: The case of the ecological ICMS in Brazil. Land Use Policy 25(4): 485–497" (Impact factor 2008: 1,821).

Building on a concise introduction to environmental federalism and associated principles of public finance, this article presents a detailed case study from Brazil. The Brazilian federal system was chosen for investigation because the country is the first worldwide to have gained practical experience with ecological fiscal transfers from state to local level. The article analyses the policy instrument "ICMS Ecológico", which was first introduced by a few states in Brazil during the 1990s (Loureiro 1998, 2002; Grieg-Gran 2000; May et al. 2002). Part of the revenue from this value-added tax is redistributed to the local level on the basis of ecological indicators. In this way, the state level uses fiscal transfers to compensate municipalities for the existence of protected areas within their territories and some other ecological services provided at the local level.

The core of this article consists in a review and analysis of the ICMS Ecológico in Brazil, presenting the different ways such an instrument may be implemented in a federally organised country. Back in 1992, the state of Paraná was the innovator state that began to adopt the ICMS Ecológico, acknowledging protected areas for watershed protection and biodiversity conservation as indicators for fiscal transfers. Since that time, 12 out of 26 states in Brazil have introduced the ecological ICMS (Figure 1.1).

Source: Ring et al. (2010a). Cartography and Geographical Information System: Kindler, Helmholtz Centre for Environmental Research – UFZ.

Figure 1.1: Brazilian states with ICMS Ecológico legislation (2008).

All twelve use conservation units as the core indicator for allocating ecological fiscal transfers; the other environment- or conservation-related indicators in use vary from state to state. Various other states that do not yet have this instrument are debating its merits and actively considering its introduction.

The indicator "conservation units" has been introduced by all states with ICMS Ecológico legislation. Conservation units are defined according to the National System of Conservation Units in Brazil and relate to the categories of protected areas for biodiversity conservation. Depending on the land-use restrictions associated with the different categories, the states choose their own conservation weights for each of the categories (Grieg-Gran 2000). In addition to taking the size of protected areas within municipal borders as a quantitative indicator, some states have implemented (much needed) quality indicators (Loureiro 2002; May et al. 2002). In this way, the existence of strictly protected and well-managed national parks gives rise to higher ecological fiscal transfers than, for example, environmental protection areas or buffer zones.

The states of Paraná, Minas Gerais, Rondônia and São Paulo have been implementing the instrument for several years. In Paraná, 50 % of all municipalities now benefit from ecological fiscal transfers; in Minas Gerais, 30 % of the municipalities receive fiscal transfers based on conservation units. Since the introduction of the instrument, both states have seen a considerable increase in protected areas at all governmental levels, although in relative terms, the incentive effect was most outstanding at the local level (May et al. 2002). This demonstrates rather nicely that instead of these protected areas being continuously perceived as obstacles to economic development, they are increasingly welcomed at the local level, not least as a source of – at times – substantial fiscal revenues. For some municipalities in rural and remote areas with low economic and agricultural productivity, ecological fiscal transfers may even represent the main revenue source.

The Brazilian experience illustrates that ecological fiscal transfers can represent both a means of compensation for land-use restrictions and an incentive to value and engage in more conservation activities at the local level. The article concludes with some reflections on the environmental policy developments now necessary in Brazil and with some comments on transferability: the positive experiences with introducing ecological indicators into the Brazilian system of intergovernmental fiscal transfers make it worth to presenting it as an innovative example that could potentially be transferred to other federal systems, not least the European Union and its member states. European decision makers are certainly interested in learning from this kind of innovative policy: Immediately upon publication, this article was selected for inclusion in the Science for Environment Policy Newsletter issued by the European Commission, the DG Environment News Alert Service (Science for Environment Policy 2008).

1.3 Applied biodiversity governance: Interdisciplinary and policy-relevant contributions

This section introduces Part III, containing chapters 6–9 of the thesis. Chapters 6 and 9 are two multi-authored articles with authors from different disciplines. Chapter 7 provides an economic perspective on nature conservation competencies in the then ongoing “Modernisation of the Federal Structure” in Germany. Chapter 8 was a contribution to a research symposium and book project, focusing on the shifting boundaries between “public” and “private” in natural resource governance.

Environmental research – often associated with the origin and development of ecology – always required the collaboration among different natural scientific disciplines (Daschkeit 1998). Interdisciplinary environmental research integrated knowledge from physics, chemistry, biology and the earth sciences. Today, it is widely recognised that scientific understanding of many aspects of global and environmental change is directly related to an understanding of human activities (e.g., Costanza 1991; Balstad Miller 1994; MA 2005a).

What is the added value of interdisciplinary research between economists and ecologists on the one hand, and between economists and other social scientists on the other? Firstly, economics benefits considerably from insights gained in collaboration with ecologists and other social scientists. Public goods and services associated with nature and biodiversity conservation are by their very nature complex; they include largely heterogeneous goods and involve, among other factors, complementary relationships among species and restricted substitutability of ecosystem functions (Jentsch et al. 2003; Naeem et al. 2009; Ring et al. 2010 a, b). Since the economic costs and ecological benefits of conservation measures are heterogeneous in space and over time, only ecological-economic research can provide policy recommendations based on complex realities (Johst et al. 2002; Wätzold et al. 2006; Wätzold et al. 2008a). Hence, economic analysis of conservation policies and instruments must build on and integrate the latest results of ecological research, if those policies and instruments are to be able to actually achieve conservation objectives.

Secondly, economics also benefits from other social sciences in that the latter provide theories and methods to understand human behaviour and perceptions – be it on an individual basis or related to social groups – that economic reasoning cannot capture (Vatn 2005). Only with this wider approach, and focusing on the formal and informal institutions of environmental decision making, is it possible to design and implement conservation policies and instruments that fit with social realities. The economically best designed compensation payments for conservation measures are useless if eligible applicants – frustrated at not being involved in the selection of large protected areas from the very beginning of the relevant planning processes – do not use them (Stoll-Kleemann 2001).

Thirdly, ecologists and other social scientists in turn benefit greatly from integrating an economic perspective into their research design. Conservation policies are still based largely on conservation biology and ecological sciences, but could have a much greater impact if economic principles and efficiency considerations were better integrated (Hampicke et al. 2006). Economics provides methodologies and instruments that help achieve given conservation goals at far lower costs; moreover, scarce resources for conservation policies last longer and achieve more conservation objectives when used cost-effectively (Wätzold and Schwerdtner 2005; Wätzold et al. 2008a, b; TEEB 2009).

Truly interdisciplinary research is often called for but still difficult to realise. Ring et al. (1999) present several stages of cross-disciplinary cooperation. The most advanced stage is integrative interdisciplinary research: Researchers from different disciplinary traditions start their collaboration "by reconceptualising and jointly defining the research problem. Interdisciplinary research often leads to reconsidering the nature of the problem in order to reach a commonly accepted problem description as a basis for subsequent project work" (Ring et al. 1999, p. 6). Although easily said, this is often ignored. Nevertheless, it provides the basis for both chapter 6 – written by ecologists and economists in the course of preparing a research agenda for integrative biodiversity research at a large research centre – and chapter 9, which analyses policies for biodiversity conflict reconciliation. Chapter 9 was written by legal scientists, an anthropologist and an economist, but builds on research results gained in close collaboration with conservation biologists and ecologists in a large EU-funded research project. Coordinated by Klaus Henle and Irene Ring (UFZ), the project involved about 60 researchers from the natural and social sciences across Europe who developed strategies for coping with conflicts between species protection and economic development, using fish-eating vertebrates and fisheries as a model case (Klenke et al. 2010).

In addition to interdisciplinary contributions, Part III includes a focus on applied and policy-relevant research in a governance context. Chapter 6 discusses the prerequisites of socially meaningful and policy-relevant biodiversity research, while chapters 7–9 are policy-relevant research contributions in themselves.

Chapters 7 (in German) and 8 (in English) both apply the concept of governance to biodiversity conservation and link it to the economic theory of environmental federalism. Theoretical arguments building on the economic theory of federalism are developed to justify in economic terms a sound distribution of conservation competencies in federal systems, serving as a scholarly basis for policy recommendations in the course of the modernisation of the federal structure in Germany (chapter 7). In this context, the spatial distribution of conservation benefits and costs deserves special attention (see also Revesz 2000; List et al. 2002; Perrings and Gadgil 2003; Wätzold et al. 2008a).

Biodiversity governance involves the coordination of policies, instruments and actors across several governmental levels, captured by the terms multi-level and multi-actor governance (chapter 8, Ring 2008c). In addition, the governance concept has to be seen in relation to different modes of environmental decision making. Decisions can be taken by governments, in which case hierarchical decision making is the rule, representing the traditional command-and-control approach to environmental policies. They may also be subject to market mechanisms, provided adequate property regimes are defined first (Vatn 2005). Last but not least, decision making may be located in the realm of civil society, represented by citizens or non-governmental organisations.

It is against this background that chapter 8 elaborates some theoretical links between the economic theory of federalism and an institutional and governance-oriented perspective on conservation policies. The chapter then proceeds to examine the design and implementation of ecological fiscal transfers in Brazil in a multi-level and multi-actor perspective. Referring to the ICMS Ecológico in Brazil, a state-driven policy instrument, the various public-private partnerships that developed with the practical implementation of the instrument are addressed, thus showing how it exemplifies hybrid forms of governance and reflects a "blurring of the sharp dichotomy between public and private in practice" (Sikor et al. 2009, p. 2).

Finally, chapter 9 combines economic, legal and anthropological expertise in an evaluation of policy mixes at different governmental levels in Europe relevant to the resolution of conflicts between species protection and fisheries. It analyses the options and limitations contained in European Union-level framework regulation aimed at resolving biodiversity-related conflicts at national and local levels. Regional case studies are drawn from Finland and Germany. Building on legal and institutional analyses at different levels of decision making, recommendations are elaborated for a better coordination of conservation policies in federal systems.

1.3.1 Biodiversity. Emerging issues for linking natural and social sciences

This chapter was published as "Jentsch, A., Wittmer, H., Jax, K., Ring, I., Henle, K. (2003). Biodiversity. Emerging issues for linking natural and social sciences. GAIA 12(2): 121–128" (Impact factor 2008: 0,653).

The article discusses new dimensions of the challenges involved in linking the natural and social sciences to conduct integrative biodiversity research. The insights presented result from the preparation of a biodiversity research plan as part of the Helmholtz Association's research programme on the "Sustainable Use of Landscapes", running from 2003–2008. To make biodiversity research both scientifically sound and policy-relevant, the authors argue, "methods and tools to link the natural and social sciences have to be adjusted to the specific issue under consideration" (Jentsch et al. 2003, p. 121).

After a brief introduction addressing the relevance of biodiversity for sustained ecosystem service provision, the article illustrates the challenges of and potential for integrative research, using as examples the key drivers of biodiversity change: disturbances, invasions, and fragmentation, as well as biodiversity conservation as a societal process and policy response to the ongoing loss of biological diversity. Disciplinary methods and tools, such as biodiversity assessment, experimental research and ecological modelling in the natural sciences, or economic valuation, policy analysis, the investigation of stakeholder perceptions and participatory methods in the social sciences (to name but a few), need to be appropriately combined in order to address the problem in its specific context.

Following this, a number of promising tools are presented which, in themselves, combine knowledge from different disciplines. One of these tools is Integrated Assessment (Horsch et al. 2001), which makes it possible to draw upon a variety of methods that integrate knowledge from both the natural and social sciences, such as ecological-economic modelling (Johst et al. 2002; Wätzold et al. 2006) and multi-criteria decision analysis (Drechsler and Burgman 2004). In a transdisciplinary research framework bridging science and society, these methods are best applied in a participatory context (Klauer et al. 2006; Rauschmayer and Risse 2005; Rauschmayer and Wittmer 2006).

Risk assessment is another methodology which – when dealing with invasive species, for example – needs to build on the biological impacts of invasive species as well as the expected social and economic consequences of biological invasions, all of which are subject to uncertainty. Spatial modelling based on Geographic Information Systems (GIS) provides an opportunity to visualise land-use changes and may serve as an important tool to demonstrate the potential effects of economic instruments and other policy measures to decision makers (e.g., Ring 2008a). Ecological-economic modelling and spatial modelling linked to planning tools are especially important in the context of increasingly fragmented landscapes, fragmentation being among the major drivers of the current loss of biodiversity (Kaule et al. 1999; Henle et al. 2004).

Research on disturbances, biological invasions and fragmentation is framed predominantly by natural scientists. This is in contrast to the field of biodiversity conservation, which is understood here as a policy response to biodiversity loss involving social processes. Human resource use can be considered a starting point of the problem, based as it is on governance structures and institutional settings, and on the rules, norms and traditions in a society (Kissling-Näf and Kern 2002; Vatn 2005). The authors encourage policy makers to adopt the perspective of the "affected actors" (Jentsch et al. 2003, p. 125). Making research relevant to stakeholders may pose a real challenge for ecologists and economists, but it is crucial for the conservation and sustainable use of biodiversity. Participatory methods involve relevant actors and stakeholder groups directly in the research process (Moll and Zander 2006) and are an important means of conducting socially relevant biodiversity research.

1.3.2 Naturschutz in der föderalen Aufgabenteilung: Zur Notwendigkeit einer Bundeskompetenz aus ökonomischer Perspektive

This chapter was published as "Ring, I. (2004). Naturschutz in der föderalen Aufgabenteilung: Zur Notwendigkeit einer Bundeskompetenz aus ökonomischer Perspektive. Natur und Landschaft 79(11): 494–500".

In October 2003, the German Parliament and the Bundesrat inaugurated a "Commission on the Modernisation of the Federal Structure" which developed proposals, among other things, to reallocate legislative competencies between the federal and the state governmental levels in Germany. The Commission itself consisted of politicians but had appointed academic experts from the juridical, economic and political sciences to provide expert knowledge and advice. Although very few of the selected experts had working experience in environmental or conservation matters, that did not prevent them from making proposals with far-reaching consequences for environmental legislation.

Since nature conservation law in Germany at the time was subject to framework regulation at the federal level in addition to 16 implementing conservation laws at the state level (*Länder*), a potential change in legislation – be it to strengthen federal competencies or to devolve competencies to the state level – might have had serious consequences for conservation law (Koch and Mechel 2004). Four experts explicitly called for the abolition of the framework regulation for nature conservation law, water resources management and some other environmental matters in combination with devolving the relevant competencies to the state level (Huber 2003; Schmidt-Jortzig 2003; Scholz 2003; Benz 2003). In contrast, Meyer (2004) favoured federal competencies for the full range of environmental and conservation laws, while Scharpf (2004) argued in a more differentiated way, considering external effects and economies of scale in environmental policy.

Against this background, the article presents economic arguments for and against nature conservation competencies at various governmental levels. Nature conservation is a public good with benefits accruing predominantly at national and international levels. The costs of conservation are distributed unequally, both in terms of economic sectors (agriculture and forestry) and in terms of the spatial distribution of protected areas across the various German *Länder*. These findings suggest the need for national-level competencies and standard setting.

The economic perspective supports keeping or even strengthening nature conservation competencies at the national level.

At the end of the Commission's work, an agreement was reached that all environmental and conservation matters should be combined in a unified Environmental Code at the federal level, consisting of several books for the different matters involved. Environmental experts have long been calling for the bundling of environmental competencies at the federal level (Haber, personal communication), as too have conservation NGOs, and now, for the first time in German history, it was finally to be realised by the Grand Coalition in the coming legislative period. Federal authorities and ministries worked under great pressure to draft the new Environmental Code. However, in early 2009, the Environmental Code failed due to the resistance of the CDU/CSU parliamentary group, prompted by CSU politicians who requested exemptions and extra regulations for the Free State of Bavaria (BMU 2009a, 2009b). Nevertheless, in July 2009 the Bundesrat finally endorsed a new environmental law reform with uniform federal requirements both for nature conservation and water legislation (BMU 2009c). However, the key element of the reform, the so-called "integrated project authorisation" (*Integrierte Vorhabengenehmigung*) will not become reality due to political resistance.

1.3.3 Biodiversity governance: Adjusting local costs and global benefits

This chapter was published as "Ring, I. (2008c). Biodiversity governance: Adjusting local costs and global benefits. In: Sikor, T. (Ed.), Public and Private in Natural Resource Governance: A False Dichotomy? Earthscan, London, pp. 107–126".

This chapter explicitly sets biodiversity conservation and policies in a governance context. Biodiversity governance is defined in terms of both multi-level and multi-actor governance. Additionally, the development and implementation of policy instruments is linked to a governance perspective on different regimes of decision making, including hierarchical decision making, market regimes, voluntary solutions, as well as mixed regimes: "Regulation, economic incentives, voluntary, informative and communicative instruments on the public side as well as activities initiated by civil society continuously interact and are of varying importance depending upon the concrete problem" (chapter 8, p. 179f.). In societal reality, policy mixes (and thus increasing interaction between public and private spheres) also apply to the field of biodiversity governance.

Furthermore, the chapter provides a link between biodiversity governance and the economic theory of fiscal federalism. Fiscal federalism is concerned with the assignment of public functions, expenditures and fiscal instruments to different levels of government. Therefore, an important link exists between multi-level governance and fiscal federalism, where the spatial characteristics of biodiversity conservation are of importance. Building on the spatial distribution of conservation benefits and costs and the need for their reconciliation to allow for sufficient provision of related goods and services, the chapter proceeds with the necessary consequences for intergovernmental fiscal relations and associated fiscal instruments.

Finally, the chapter is devoted to a case study focused on the Brazilian ICMS Ecológico as an instrument capable of reconciling the local costs and spillover benefits of biodiversity conservation. Although a hierarchical, state-governed instrument, it is an economic instrument by nature. Furthermore, the ICMS Ecológico emerged as an economic incentive for new forms of public-private interaction. Hence, the instrument and its impacts are a good illustration “of the shifting boundaries between notions of publics and privates in various aspects” (chapter 8, p. 181).

1.3.4 Protected species in conflict with fisheries: The interplay between European and national regulation

This chapter was published as “Similä, J., Thum, R., Varjopuro, R., Ring, I. (2006). Protected species in conflict with fisheries: The interplay between European and national regulation. Journal of European Environmental and Planning Law 3(5): 432–445”.

The article is based on interdisciplinary and applied research results from the EU-funded 5th framework programme research project “FRAP – Development of a procedural Framework for Action Plans to Reconcile conflicts between the conservation of large vertebrates and the use of biological resources: Fisheries and fish-eating vertebrates as a model case”. The project was coordinated jointly by the UFZ Departments of Conservation Biology (K. Henle) and Economics (I. Ring), with further partners from Finland, Sweden, Denmark, Portugal, Italy, Austria, the Czech Republic and the UK. In each of the selected model regions across Europe an interdisciplinary team consisting of natural and social scientists performed empirical and participatory research on conflicts between fisheries (or aquaculture, depending on location) and protected species: the species investigated were the Eurasian otter (*Lutra lutra*) in Central Europe (Germany, Czech Republic and Austria) and Portugal, the grey seal (*Halichoerus grypus*) in Finland and Sweden, and the great cormorant (*Phalacrocorax carbo sinensis*) in Denmark, Italy, and Germany (Klenke et al. 2010).

The project’s focus was on a problem that is encountered worldwide (e.g., Conover 2002; Woodroffe et al. 2005; Manfredo et al. 2009): Successful public conservation policies at various governmental levels have increased some populations of protected species to the extent that they are causing damage to human activities including agriculture, forestry, fisheries and aquaculture. As a response to growing biodiversity conflicts, public authorities and all parties interested in species conservation need to develop strategies for conflict resolution (Treves et al. 2006; Ring 2009; White et al. 2009). Otherwise they once again risk species populations dwindling, largely for the same reasons why these populations decreased in the past: species in conflict with human activities were often hunted by humans who were enduring damage (or thought they were), or the species were hunted down and killed for other reasons; or – at the very least – they were constricted in their habitat requirements.

The article itself – as part of the EU project’s work package 4 (Legal and institutional framework) – explores the options offered by European regulation and the restrictions it imposes on European Member States. Relevant European policy fields are the Common

Fisheries Policy, nature conservation regulation (with the Habitats and the Birds Directive), EU state aid regulation aimed at avoiding distorted competition within the Common Market, and European funds providing financial support to Member States. The interrelationship between European and national regulation is investigated on the basis of experience with German and Finnish biodiversity reconciliation policies.

Finland and Germany have both developed reconciliation policies for the conflicts between nature conservation and fisheries. In Finland, the major conflict relates to grey seals, which cause damage to small-scale coastal fisheries. In Saxony, Germany, cormorants and otters are causing damage to aquaculture by feeding on carp. Reconciliation policies in both countries include a mix of policy instruments and measures including the management of populations, protective hunting, support for technical measures and various types of compensation payments. All these measures are affected by European policy and law, though no special reconciliation policy has been adopted at European level. The article concludes with suggestions for better coordination of reconciliation policies between different governmental levels.

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Part II:

Integrating local ecological services into intergovernmental fiscal transfers

2 Ecological public functions and fiscal equalisation at the local level in Germany

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Abstract

The concept of sustainability requires the acknowledgement and appropriate financing of ecological, economic and social public functions at all levels of government. Whereas the significance of socio-economic functions has a comparably long tradition in federal systems, the respective consideration of ecological functions is still waiting to be fully realised. The aim of this article is to analyse the roles fiscal federalism and fiscal instruments can play in addressing ecological public functions assigned to decentralised jurisdictions in federal systems. For this purpose, an investigation of intergovernmental fiscal relations and their significance for the local level is carried out by way of example for the Federal Republic of Germany. Based on a status quo analysis of the various German fiscal equalisation laws at the local level, the article presents options for systematically integrating neglected ecological functions and indicators into the existing legal framework. The empirical analysis shows that end-of-the-pipe and infrastructure-related ecological functions are already strongly integrated by way of conditional grants. However, precautionary functions such as nature and resource conservation issues are still largely absent. Therefore, a comprehensive consideration of ecological public functions in Germany calls for the integration of nature conservation issues and respective indicators into the system of intergovernmental fiscal relations.

Keywords:

Environmental federalism, environmental sustainability, intergovernmental fiscal relations, fiscal grants, local jurisdictions, Germany

2.1 Introduction

Sustainable development has become a key concept within environmental policy analysis. When adequately implemented, it adds new elements and criteria to the formulation and evaluation of environmental policy (van den Bergh 1996). In its early stages, it developed within the context of global environmental problems (e.g. WCED 1987). However, sustainable development involves the proper integration of environmental concerns into policy-making at all levels of decision (Hardy and Lloyd 1994). Its implementation is a task of not just environmental institutions but also economic, financial, agricultural and social institutions (van den Bergh 1996). Based on the concrete problems in mind, the policy levels and institutions which have to be addressed are international, national, regional and local. Linking these thoughts to economic reasoning, the assignment of functions to different levels of government is found to be part of the basic theory of fiscal federalism (Oates 1972, 1999). Therefore, there is an important link between the implementation of the sustainability concept, environmental policy, and the economic theory of federalism.

In the following, special attention will be paid to the significance of decentralised levels of government for achieving environmental sustainability, such as the local level (i.e. the municipal or district level) or the federal state level as an intermediate level below the nation states (e.g. the German *Länder* or the cantons in Switzerland). These administrative levels are of prime importance for the regional approach to sustainable development that has previously received comparatively little attention (van den Bergh 1996), even though growing interest can be noted (Ring et al. 1999; Köhn et al. 2001).

Due to the public-good nature of many environmental goods and services, an essential step to achieve environmental sustainability at the regional level depends on the realisation and success of conservation and environmental policies at decentralised jurisdictions. A central problem, however, relates to the lack of financial resources for these issues compared to other, more traditional and established fields of socio-economic activities. Although ecological public functions may be well defined for the various administrative levels in respective laws and regulations, the resources in terms of money and personnel necessary to ensure that jurisdictions carry out these functions in a qualified manner are often lacking. This is especially true for conservation-oriented tasks (Henle 1995). Consequently, considerable shortcomings in the implementation process of relevant policies are almost inevitable (Kneer 1999). To secure the adequate provision of environmental goods and services at decentralised levels of government, fiscal grants as part of fiscal equalisation at the local level have been increasingly discussed for Germany (SRU 1996, 2000; Rose 1999; Ewringmann and Bergmann 2000; Ring 2000, 2001a). Initial initiatives towards integrating biodiversity into intergovernmental fiscal transfers are to be noted for Switzerland (Köllner et al. 2002).

The following contribution is devoted to this kind of fiscal instrument, using the concept of sustainability as a normative basis and guiding principle to design intergovernmental fiscal relations in federal states. For this purpose, a brief review of basic ideas and principles of environmental federalism will provide the groundwork for further analysis. Following a definition of ecological public functions, their significance and binding nature in legal terms will be presented within the context of the European Union. An empirical investigation of ecological functions in intergovernmental fiscal relations is then carried out by way of example for the Federal Republic of Germany. Specifically, the various fiscal equalisation laws of the German states are proposed as a means to integrate ecological aspects and thus help decentralised jurisdictions to sustain their natural endowment by offering compensation for their ecological services.

2.2 Environmental federalism and ecological public functions

2.2.1 Fiscal federalism and the environment

Fiscal federalism as a subfield of public finance explores the roles of the different levels of government and the ways in which they are interrelated through fiscal instruments. It aims at improving the performance of the public sectors and the provision of their services by aligning responsibilities and fiscal instruments with the proper level of government. Environmental federalism links environmental issues with the basic theory of fiscal federalism. The public-good nature of most environmental goods and services raises the question of which governmental level should best be responsible for their provision. Following the general decentralisation rule for the allocation function of public services (Musgrave 1959; Oates 1972), lower levels of government should be assigned the task of providing environmental goods and services where appropriate. This way allows the regionally differing preferences of the population to be reflected more adequately (Tiebout 1956).

The general decentralisation rule for allocating public goods and services only applies in the absence of economies of scale. In the presence of economies of scale, the provision of the public goods concerned should be moved to the adequate, cost-efficient centralised level (Postlep and Döring 1996). Furthermore, due to the characteristics of non-rivalry and non-excludability of many public goods, some of them are associated with spatial externalities between jurisdictions (spillovers). Here, the principle of fiscal equivalence advocates achieving a “match between those who receive the benefits of a collective good and those who pay for it” (Olson 1969). Environmental federalism necessitates an additional element concerning the principle of fiscal equivalence. It is no longer only the group benefiting that should come up with the costs of the public goods and services produced, and consequently, decide on the size of the jurisdiction; it is also the individuals responsible for polluting the environment who must be considered when deciding on the boundaries of the appropriate jurisdiction. To sum up this context, the term “ecological equivalence” has been coined (Huckestein 1993). Döring and Fromm (1997) developed a two-step procedure to apply the principle of fiscal equivalence to environmental problems. Firstly, the appropriate jurisdiction of the environmental problem concerned has to be determined, including all the parties affected (e.g. polluting and harmed individuals). Secondly, a decision on financing public intervention has to be made. This decision may follow the polluter-pays principle, but it can also be made on the grounds of other principles (e.g. Coase theorem).

Due to the specific characteristics of natural resources and environmental quality, the implementation of the general decentralisation rule begs a differentiated view. This is reflected in the ongoing debate on the responsibilities of the national or even supranational level versus the state or local governmental level in setting environmental standards or carrying out other environmental functions (Döring 1997; Scheberle 1997; Oates 1999).

In the United States, far-reaching proposals have been put forward under the banner of "devolution" to return environmental standard-setting and policy-making from the national level to local and state governments and even, wherever possible, to private individuals (Anderson and Hill 1996). By contrast, Smith et al. (1997) highlight the importance of the information necessary, and the relevance of specific details to the application of conceptual models of federalism that "are especially important in structuring the jurisdictional responsibilities for the design and implementation of different aspects of environmental policy". The spatial dimension of environmental resources, particularly their linkages in creating spillover effects, must be considered when evaluating the costs of attaining any ambient quality level. Environmental goods require the production component of the delivery of local public goods to be more carefully investigated. This is a difference compared to most discussions in fiscal federalism that more often focus on available alternatives for financing public intervention or on preference heterogeneity in its implication for the amounts and types of public goods provided by different jurisdictions (Smith et al. 1997). For Canada, a critical view of passing too much responsibility from the federal to the provincial level is presented by Harrison (1996).

In Europe, environmental federalism has been rediscovered and extensively discussed since the Treaty of Maastricht on European Union of 1992 that fundamentally strengthened the principle of subsidiarity (Huckestein 1993; Hansjürgens 1996; Döring 1997; Oates 1998). According to this principle, public policy and its implementation should be allocated to the smallest jurisdiction with the competence to achieve the objectives. Despite the firm and explicit reinforcement of the subsidiarity principle in the new Article 3b of the Treaty, a fair amount of leeway is left for interpretation. Therefore, any concrete implementation of environmental policy has to go into the specific detail of the matter concerned. For example, the discovery and dissemination of basic knowledge about environmental harm and about the effectiveness of various policy instruments need to be assigned to a more centralised level of government, for this kind of public good tends to be underprovided at decentralised levels (Oates 1998). The same holds for highly mobile environmental compartments and associated pollutants that easily cross national boundaries, creating far-reaching spatial externalities (Hansjürgens 1996). Sulphur dioxide, carbon dioxide and ozone are typical air pollutants with a need for emission policies subject to more centralised governmental levels.

In contrast, environmental policy associated with less mobile environmental compartments is better suited for assignment to lower levels of government. This is due to the lower probability of causing cross-boundary spatial externalities. Problems of land use and soil contamination, as well as functions associated with inland waters, such as lakes or groundwater resources, can usually be solved within national boundaries. The same holds for many issues related to nature conservation and landscape protection. Depending on the type of natural resources and environmental problems, there is room for even further decentralisation. In the federal system of Germany, environmental functions related to land-use planning, water resources and nature conservation are only subject to framework regulation at the national level. Practical implementation is delegated to the various state laws, regulating the respective functions of the state, regional and local levels of government.

Despite the general suitability of some functions to be assigned to lower governmental levels, spatial externalities may also exist requiring appropriate solutions. This is the case for special priority areas, e.g. regarding the conservation of rare species or the protection of drinking-water resources, that cause costs within the concerned jurisdiction, but externally benefit others. Although developed for other public goods, such as education, a solution to this kind of problem has already been suggested by Olson (1969). Provided diseconomies of large-scale operation call for local provision, these externalities should be internalised through government grants from more centralised levels, compensating local government for the external benefits of its expenditures.

2.2.2 The significance of ecological public functions

In the literature on environmental federalism, more general and theoretical expositions on linking federalism and the environment are often made for the case of environmental pollution. To stress the importance of natural resources protection on the one hand, and environmental pollution on the other, the use of the more comprehensive adjective "ecological" instead of "environmental" is introduced here for the respective public functions. The term "ecological public functions" is also used with reference to the three dimensions of the concept of sustainability, explicitly indicating the need to consider ecological, economic and social public functions in intergovernmental fiscal relations.

Concerning concrete areas of public activities, ecological public functions consist of the protection and sustainable use of natural resources, living organisms, ecosystems and landscapes. These precautionary-type functions also comprise activities aiming at the conservation of nature as a sound living basis for human life, including recreational purposes. Furthermore, ecological public functions address negative effects of human activities on the environment, including environmental pollution in the form of emissions, waste and contaminated sites, but also impaired or destroyed landscapes. Consequently, a comprehensive analysis of ecological public functions within intergovernmental fiscal relations should highlight precautionary functions referring to areas such as soil, water, nature protection, landscape preservation and recreation. It should also look at public functions associated with aftercare, such as sewage and waste disposal or the rehabilitation of contaminated sites and landscapes. The realisation of the concept of sustainability calls for the consideration and appropriate financing of these ecological public functions at any governmental level. With reference to the following case study, there are basically two arguments on which this claim can be legally founded within the European context.

The first argument is connected to a country's membership in the European Union and the binding force of its legal framework. Although a few European initiatives can be traced back to the 1970s, the first comprehensive approach to environmental policy consisted of the formulation of the First Environmental Action Programme in 1983. Since the Single European Act of 1987, a firm legal basis for a common environmental policy was created in the form of Articles 130 r–t of the European Treaties (Döring 1997). Finally, the Treaty of Amsterdam on European Union (1997) established the principle of sustainable development as part of the European legal framework. Firstly, sustainable development is part of the Treaty's preamble as a general principle to be pursued by the Member States. Secondly, in the new Article 2 of the Treaty of Amsterdam, the implementation of sustainable development expressly enjoys equal rights with the promotion of economic and social progress, and the achievement of high employment levels (Frenz and Unnerstall 1999). The principle of sustainable development has hence become a leading concept for policy formulation in the Member States, and in combination with the subsidiarity principle, is waiting to be implemented at the appropriate governmental levels.

The second argument is connected to the development of conservation and environmental policies at national levels. As of the 1970s, in many European countries including Germany, a sophisticated and complex legal framework came into force that assigned numerous functions to different governmental levels. Depending on the design and interrelationship among federal, state, regional and local ecological functions, respective public expenditures accrue. However, as opposed to other public functions existing for many decades and endowed with comparatively substantial financial resources to secure the provision of the related public goods and services, nature conservation and environmental policy still suffer from a lack of financial resources due to their short history and the relatively weak influence of environmental interest groups in the political process (e.g. for Germany, Soell 1989; Kaule 1991).

2.2.3 Fiscal federalism co-evolving with environmental sustainability

Both the general principle of sustainable development as adopted at the European level of government and the numerous ecological functions already assigned to the various governmental levels within nation states call for consideration of ecological functions in intergovernmental fiscal relations. Obviously, both arguments rest on the normative assumption that current legal frameworks, be it the European Treaties or the existing set of rules concerning national conservation and environmental policy, are accepted as an orientation for adjusting intergovernmental fiscal relations.

This assumption can be seen to deviate from standard economic studies in fiscal federalism. To determine any optimum level of government for the provision of public goods and services, investigations usually abstract from a given legal framework (Hansjürgens 1996). By contrast, economic reasoning in the public finance literature aims at identifying the optimal size of the jurisdiction from the angle of economic efficiency. Although this is an important standpoint, the primary interest of this article lies in a different direction.

Here, we start from the basic viewpoint of sustainable development as a concept necessary to be implemented, and seek possible ways of its realisation. Furthermore, although we are well aware of the shortcomings and the room for improvement of the present legal frameworks on nature conservation and the environment, for the purpose of this study we take it as a given fact that ecological functions are assigned to different governmental levels. Our concern is instead to contend that intergovernmental fiscal relations in federal systems have not yet adequately acknowledged and integrated the development achieved by contemporary societies in the field of conservation and environmental policy during the last three decades. It is argued that fiscal relations and fiscal instruments should mirror societal innovations, coevolve with the concept of sustainability, and specifically, help to implement environmental sustainability (Ring 1997). In our case this means that the relevant regulations should also consider ecological public functions just as they consider social and economic public functions.

In recent years, a variety of suggestions for the greening of government grants within the system of intergovernmental fiscal relations has been put forward for Germany, concentrating on fiscal equalisation at the local level. Therefore, the German federal system is chosen as an example for further empirical analysis, presenting the status quo of and further perspectives for considering ecological public functions in fiscal grants as relevant for decentralised levels of government.

2.3 Fiscal equalisation at the local level in Germany

2.3.1 Fiscal grants in the German federal system

Intergovernmental fiscal grants play a distinctive and important role in fiscal federalism that can serve a number of different functions. Firstly, the literature emphasises the role of the internalisation of spillover benefits to other jurisdictions. Secondly, based on normative considerations of equity, they serve the purpose of fiscal equalisation among different jurisdictions. These equalising grants play a major role in the fiscal system of Germany, as well as in other federal systems such as Canada and Australia (Oates 1999).

In the German federal system, basic regulations concerning intergovernmental fiscal relations are part of the German Constitution. It was the financial reform of 1969 that fundamentally determined the present system of fiscal federalism (Lenk 1993; Zimmermann and Henke 1994). The distribution of responsibilities among the federal, state and local level follows the principle of subsidiarity. The federal level carries out tasks of a general, usually nationwide, character. State and local authorities are politically responsible for regional and local development. Federal and state governments draw up their own budgets and bear individual responsibility for their implementation. In addition, there are some supranational tasks that are planned and financed jointly by the federal government and the states. These “joint tasks” relate to improvements in the regional structure of the economy and the structure of agriculture, as well as the protection of the coastline or the construction of university buildings (Seidel and Vesper 1999).

The German system of intergovernmental fiscal relations is based on two major pillars. On the one hand, the fiscal equalisation law (*Finanzausgleichsgesetz*) in its version of 1969, together with its numerous subsequent amendments, specifically lays down the intergovernmental fiscal relations between the central level and the states (*Länder*). Essentially, they cover the vertical fiscal relations between the federal level and the state level, and horizontal fiscal relations among the states themselves. On the other hand, there are intergovernmental fiscal relations within the states that are regulated in the 13 different fiscal equalisation laws of the various German *Länder* addressing fiscal equalisation at the local level (*Kommunaler Finanzausgleich*). The latter comprises vertical fiscal relations between each state level and its local level of government (local and district councils), and in some of the states, horizontal fiscal relations between the local councils themselves. Due to the special importance of local jurisdictions for implementing environmental sustainability at the regional level, further analysis will focus on fiscal equalisation at the local level.

The local government level is given a relatively high degree of autonomy by the German Constitution. District councils and local councils (cities and communities) derive their authority by implementing state or federal legislation, or by managing areas for which they have sole and direct responsibility. To carry out their functions, local governments require specific fiscal instruments (Zimmermann 1999). For this purpose, they rely on their own revenue sources such as local taxes and charges. In addition, local governments gain a considerable amount of income based on the intergovernmental fiscal relations laid down in the various fiscal equalisation laws. As for the vertical flow from the state to the local level, intergovernmental fiscal grants represent an important source of local income in Germany. More than 50 % of average local income in eastern Germany (represented by the German *Länder* Brandenburg, Mecklenburg-Western Pomerania, Saxony, Saxony-Anhalt and Thuringia) is gained from fiscal equalisation at the local level. In West Germany (Baden-Württemberg, Bavaria, Hesse, Lower Saxony, North Rhine-Westphalia, Rhineland-Palatinate, Saarland and Schleswig-Holstein), own revenues are higher, but grants from fiscal equalisation still represent almost 30 % of average local income (Karrenberg and Münstermann 2000).

In a number of German states, the greater share of vertical transfers is given in the form of "unconditional grants", i.e. lump-sum transfers to the local level of government to be used in any way the recipient wishes. The remaining share of vertical transfers is "conditional", i.e. it is given for specific purposes and partly in the form of matching grants (the grantor only finances a specified share of the recipient's expenditure).

2.3.2 The role of socio-economic functions and indicators

Intergovernmental fiscal grants play an important role in local and regional development by way of securing financial resources to local jurisdictions to carry out their various public functions. Nevertheless, in their current mode of implementation as set out in the fiscal equalisation laws of the German states, they also represent a source of inequality between urban and rural areas.

In the German *Länder*, lump-sum grants are assigned on the basis of the fiscal need of a local jurisdiction in relation to its fiscal capacity (revenues based on local taxes, etc.). The fiscal need is determined by adding up the "principal" and "additional" approaches. Whereas the principal approach has to take into account the general and average fiscal need of the jurisdiction, additional approaches are supposed to consider extra community-individual, regional and transregional financial burdens.

In most German states, the principal approach for allocating lump-sum transfers is defined by the product of the number of inhabitants and a weighting factor that increases with the population. Accordingly, the more inhabitants a local jurisdiction has, the higher the lump-sum transfers. The basic argument for this relationship dates back to Brecht (1932), who formulated the "law of the progressive parallel connection between public expenditure and population concentration". Further justification has been put forward by Popitz (1932), who argued that urban populations actually have a higher demand for public goods than rural ones. Nowadays, both arguments are highly controversial, and in the states of Rhineland-Palatinate and Schleswig-Holstein the weighting factor has already been abolished. For other states, the weighting factor is subject to much criticism in the public finance literature (Lenk and Birke 1998). However, even in the absence of the weighting factor, the dominance of an inhabitant-based indicator generally favours urban areas as opposed to rural areas due to the lower population densities of the latter.

Complementary to the inhabitant-based principal approach, some of the states also have additional approaches for allocating lump-sum transfers, e.g. taking into account the number of schoolchildren, the social burdens, or the central functions of the local jurisdictions. Usually, these additional indicators are also related to the socio-economic public functions of the jurisdictions.

Concerning conditional grants for specific purposes, once again great consideration of socio-economic functions is to be observed in the various fiscal equalisation laws of the German *Länder*. Here, areas such as transport and road construction, social burdens (e.g. assistance to the unemployed or the poor), health services, education and cultural investments (allocations to theatres and opera houses) are explicitly addressed as a potential motivation for application.

To sum up, urban and densely populated areas presently benefit from the system of intergovernmental fiscal relations due to their socio-economic and cultural functions and the respective indicators, especially those on which lump-sum transfers are based. Conversely, rural areas, due to their low population densities, receive a much smaller share of the overall vertical flow. However, rural and remote areas usually provide ecological services for society as a whole, such as drinking-water protection, resource provision, the availability of unfragmented landscapes that are valuable habitats for endangered species, and recreational purposes. Therefore, successful sustainable development strategies have to address both these areas; in fact, they have to consider the imbalance between urban and rural areas concerning the specific land uses in place (SRU 1996; Ewers et al. 1997; Ring 2001b).

2.4 Ecological public functions at the local level: Status quo and perspectives for Germany

Ecological public functions are already part of fiscal equalisation at the local level in a variety of different modes (Ring 2001a). However, before presenting the empirical results based on an analysis of all fiscal equalisation laws at the local level as currently in force for the states of the Federal Republic of Germany,[1] some preliminary remarks must be made. It must be borne in mind that due to the federal structure in Germany, all German states are familiar with a great number of additional earmarked grants, mostly in the form of incentive programmes for environmental or conservation-oriented purposes (e.g. for Saxony, Frank and Ring 1999). These programmes can be exclusively implemented at the state level; some of them are jointly managed by the federal and the state level, while others are combined with agri-environmental programmes at the European level (Plankl 1996). A comprehensive analysis of earmarked grants related to ecological public functions should ideally consider all additional regulations alongside fiscal equalisation, where local jurisdictions can serve as applicants. The main purpose of this analysis, though, is to precisely elaborate the present status of ecological public functions within fiscal equalisation at the local level in Germany. This is due to the fact that grants from fiscal equalisation are among the most important income sources for local jurisdictions in Germany. Furthermore, fiscal equalisation serves a number of additional purposes from the angle of public finance. On the one hand, fiscal grants do not necessarily have to be earmarked and can still be based on ecological indicators referring to the fiscal need for ecological public functions. On the other hand, spillover effects related to ecological public functions can be effectively addressed within the system of intergovernmental fiscal relations just as spillover effects are currently addressed with respect to socio-economic and cultural functions of urban agglomerations. Therefore, the following analysis of fiscal equalisation at the local level is designed firstly to identify the kinds of ecological functions considered (respectively neglected), and secondly to point out the types of grants and indicators currently used for ecological functions. Based on the results, recommendations can be made for the comprehensive consideration of ecological public functions within fiscal equalisation at the local level in Germany.

2.4.1 Area-related indicators as an indirect approach

In a very basic sense, area-related approaches constitute a first step towards acknowledging ecological functions (Ring 2000). This is due to the importance of area and its associated land uses for many ecological functions. The need to consider area as an indicator for allocating intergovernmental fiscal grants is especially important for large communities and for district councils as opposed to smaller communities that are part of the wider district (local councils) or district-independent cities. The larger a community or district is, and the farther away it is from an urban agglomeration, the higher the relevance of area-related indicators. These re-

[1] Only the city states Berlin, Bremen and Hamburg are excluded. Due to their specific situation, local and state public functions can hardly be separated and so no fiscal equalisation laws exist.

mote areas are often characterised by low population densities and, in terms of land use, by a higher proportion of agricultural land and forestry, as well as more valuable habitats for rare species. Therefore, it is not surprising that consideration of area-related indicators has long been championed by the German Association of District Councils (Henneke 2001). This is due to the cost relevance of area for financing specific local public functions. The larger the community or district area in combination with a lower population density, the more expensive the production of certain public goods and services. Bergmann (1999) and Henneke (2001) mention additional costs related to both ecological and socio-economic public functions such as nature conservation, agricultural affairs, waste disposal, water supply, public transport, education and health services.

Therefore, the majority of fiscal equalisation laws at the local level already consider area-related indicators in one way or another, except in Lower Saxony. In Brandenburg, Mecklenburg-Western Pomerania, Rhineland-Palatinate and Saxony-Anhalt, area is one of a number of indicators for the distribution of lump-sum transfers. Other states use area as one indicator to distribute lump-sum transfers for municipal investments. Most states make use of conditional grants for selected public functions that become more expensive with a larger district area. However, these earmarked grants usually cover road construction and maintenance, public transport and transport for schoolchildren. Although theoretical claims clearly cover ecological functions, existing regulations concerning area as an indicator in the various fiscal equalisation laws at the local level predominantly concentrate on socio-economic functions.

2.4.2 The direct consideration of ecological public functions

Beyond area as a rather indirect indicator, most fiscal equalisation laws at the local level also directly consider ecological functions. Firstly, the total amount of finance available for distribution to local government can be divided such that ecological functions are also taken into account. Before any indicators come into play, a certain amount of funds can be earmarked in advance for selected ecological purposes in addition to the monies set aside for traditional purposes (Ewers et al. 1997; Rose 1999). The states of Bavaria, Baden-Württemberg, Saxony and Mecklenburg-Western Pomerania make use of this possibility (Ring 2001a).

Secondly, additional approaches can include ecological functions as a basis for calculating the fiscal need in the course of determining lump-sum transfers. For example, Saarland has an additional approach for local jurisdictions suffering from mining damage; Hesse, Saarland and Rhineland-Palatinate use them for local public functions related to spas.

Thirdly – and this is by far the option most used – fiscal equalisation at the local level considers ecological functions by means of conditional grants (Ring 2001a). Table 2.1 gives an overview of supported areas and measures in the states concerned. Most of the fiscal equalisation laws explicitly specify measures related to sewage disposal, water supply and waste disposal. Conditional grants for the prospecting and remediation of contaminated sites are also quite common. Further ecological functions can only be sporadically found in a few fiscal equalisation laws, mainly concerning precautionary-type measures related to nature and resource protection or landscape conservation. Brandenburg places emphasis on functions related to agriculture and tourism, while Mecklenburg-Western Pomerania considers landscape maintenance. In the way of conditional grants, Hesse supports projects in the areas of biotope protection, biotope networks and water conservation schemes. North Rhine-Westphalia specifies grants for the preservation of cultural landscapes and natural monuments, including the ecological rehabilitation and landscaping of the Emscher-Lippe area.

To sum up, there is a widespread tendency to support end-of-the-pipe infrastructure such as sewage and waste disposal. With the exception of functions related to (drinking) water, resources protection and nature conservation activities are rarely supported. Furthermore, most ecological functions implemented in fiscal equalisation laws are only represented by conditional grants or the provision of loans for local government. Apart from area as an indirect indicator for certain ecological functions, there is no principal or additional approach based on an indicator generally taking into account ecological functions comparable to the consideration of inhabitants for the socio-economic functions. Consequently, ecological functions predominantly provided by rural and remote areas are underrepresented in fiscal equalisation at the local level, and therefore the respective jurisdictions are not compensated for the external benefits of their expenditure. This insufficient spatial coincidence of costs and benefits may lead to an underprovision of the public goods and services concerned (Bergmann 1999).

Table 2.1: Ecological functions in fiscal equalisation at the local level in Germany

Area	Supported measures	German states
Soil	Prospecting and remediation of contaminated sites, recultivation	BY, BW, HE, NW, TH
Water	Water protection	HE
	Water supply	BY, BW, HE, MV, RP, SL, SN, TH
	Sewage disposal	BY, BB, BW, HE, MV, NW, RP, SL, SN, TH
Nature conservation	Nature protection and landscape conservation	BB, HE, MV, NW
Recreation	Spas	BW, NW, RP
	Recreation and tourism	BB, MV, RP, SH
Waste disposal	Waste disposal plants	BY, HE, MV, RP, SL, TH
Energy	Energy saving measures	HE

BW, Baden-Württemberg; BY, Bavaria; BB, Brandenburg; HE, Hesse; MV, Mecklenburg-Western Pomerania; NW, North Rhine-Westphalia; RP, Rhineland-Palatinate; SL, Saarland; SN, Saxony; SH, Schleswig-Holstein; TH, Thuringia.

Source: Ring 2001a.

2.4.3 Fiscal options for further considering ecological functions

One way of counteracting the underprovision of ecological goods and services would be to systematically integrate ecological functions into the various fiscal equalisation laws at the local level. Concerning the types of ecological functions, precautionary and intergenerational ecological functions must be stressed, such as nature conservation and landscape preservation as well as soil and water protection. With regard to the types of grants, a distinction has to be drawn between lump-sum and conditional grants.

There are two options for considering ecological indicators in the course of determining lump-sum grants in Germany. In addition to existing inhabitant-based principal approaches, the calculation of the general fiscal need of a local jurisdiction may also be based on appropriate ecological indicators. Complementary additional approaches might be adapted to take into account specific ecological needs. Some authors favour extending the additional approaches, since they are explicitly meant to unfold a certain steering function (Bergmann 1999).

However, only the category of direct costs accruing for ecological public functions provides a basis for extending both principal and additional approaches. Although a lack of financial resources for conservation-oriented tasks is often stated, there are few empirical studies that actually try to determine budgetary deficits related to these tasks. This is even more true for the corresponding costs of local jurisdictions (an exemption is Karl (1995)). Thus, there is a need for empirical studies investigating the direct costs borne by local governments due to the provision of ecological goods and services. Implementation deficits due to the lack of resources in terms of personnel or monies available should also be considered. Founded on these studies, appropriate abiotic or biotic indicators need to be identified that might constitute a link between the ecological functions and the respective costs required for their provision (Ring 2000).

The category of conditional grants is especially well suited to internalise spillover effects (Oates 1972; Fischer 1988). Local governments bear financial burdens for providing ecological goods and services that at least partially benefit other jurisdictions. This burden can accrue in the form of direct costs, e.g. for conserving protected sites of (inter)national importance. It may also show up in the form of opportunity costs, for a protected area status may restrict other (economic) development options. Ideally, the grantor finances the share of the recipient's expenditure that externally benefits other jurisdictions. However, due to the well-known difficulties of accurately determining spillover effects, appropriate procedures have to be developed to approximately decide the local and external benefits of relevant expenditures. The local benefit for providing certain ecological goods and services should ultimately be reflected in the matching contribution of the local government to receive a conditional grant.

Furthermore, the category of conditional grants may be used to adjust local behaviour to the objectives of centralised governments. Conditional grants can thus be introduced as an incentive for local governments to spend more money on conservation-oriented tasks. Although this option has already been realised in some of the fiscal equalisation laws, this kind of steering effect need not necessarily be exerted by fiscal grants within the system of fiscal equalisation at the local level. There are a number of earmarked grants as part of conservation-oriented incentive programmes in the German states that can also be used for this purpose.

2.5 Conclusion

Sustainable development requires the acknowledgement and financing of ecological, economic and social public functions at all levels of government. Whereas economic and social public functions have a rather long tradition in the intergovernmental fiscal relations of federal systems, ecological public functions have only appeared comparatively recently on the political agenda. An important step towards implementing sustainable development thus consists in strengthening ecological functions. Well-established economic, financial and social institutions such as the fiscal transfer systems in federal states presently face the challenge of co-evolving with the implementation of environmental sustainability at various governmental levels. Finally, ecological aspects will have to be regarded as a matter of course in all relevant domains of intergovernmental fiscal relations.

Although this article presented a national case study, the general message can be transferred to other federal systems. The binding force of the European legal framework with its common environmental policy and the adoption of the principle of sustainability clearly give a direction for the greening of intergovernmental fiscal relations in all Member States. Furthermore, many other federal systems around the world have committed themselves to comply with international environmental treaties and to implement the concept of sustainability. Sophisticated legal frameworks now exist in numerous states assigning ecological functions to different governmental levels. However, the kinds of recommendations to be made for considering ecological public functions at various levels of government strongly depend on the type of federal system investigated (e.g. co-operative versus competitive federalism), the general role and functions of different jurisdictions within these systems, and the specific environmental legislation in force.

Research at the interface of implementing the sustainability concept, conservation and environmental policies, and the economic theory of federalism is still more or less in its infancy. Few studies exist so far that investigate intergovernmental fiscal relations for their potential to adequately consider ecological aspects in terms of public functions and appropriate financing. The German case study has shown that although ecological functions are partly considered within the system of fiscal equalisation at the local level, there is still a long way to go before intergovernmental fiscal relations systematically address ecological functions. The empirical results seem to somewhat mirror the general historical development of environmental policies. In the early phases of environmental policy, a narrow focus on the point of emissions favoured technological, end-of-the-pipe solutions to acute and mostly local environmental problems. In later and more mature phases of environmental policy, the importance of precautionary, long-term and co-evolutionary approaches was highlighted, calling for integrative concepts (Ring 1997).

Both aspects have to be carefully considered in the course of analysing intergovernmental fiscal relations. Environmental federalism would then be achieved regarding both the ecological functions and the financial resources needed to secure the provision of ecological services. In this way, a basic prerequisite for sustainable development – taking into consideration ecological, economic and social aspects – would be fulfilled for fiscal instruments, thereby helping to translate sustainability into reality.

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3 Ökologische Aufgaben und ihre Berücksichtigung im kommunalen Finanzausgleich

Ecological public functions and their consideration in fiscal equalisation at the local level

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Abstract

Regionale Nachhaltigkeit bedingt die Wahrnehmung und Finanzierung ökologischer, ökonomischer und sozialer öffentlicher Aufgaben durch kommunale Gebietskörperschaften. Während die Bedeutung sozioökonomischer Aufgaben im föderalen System der Bundesrepublik Deutschland seit langem anerkannt ist, werden ökologische Aufgaben bislang nur teilweise berücksichtigt. Auf der Basis einer empirischen Analyse der kommunalen Finanzausgleichsgesetze der 13 Flächenländer präsentiert Irene Ring den Stand der Berücksichtigung ökologischer Aufgaben und entwickelt Vorschläge für deren systematische Integration.

3.1 Einführung

Nachhaltige Entwicklung erfordert die Berücksichtigung von Umweltbelangen auf allen politischen Entscheidungsebenen. Aus einem ökonomischen Blickwinkel betrachtet, findet man die geeignete Zuweisung von öffentlichen Aufgaben zu unterschiedlichen föderalen Entscheidungsebenen als Gegenstand der ökonomischen Theorie des Föderalismus. Aus diesem Grund stehen die Operationalisierung des Leitbildes der Nachhaltigkeit, die erfolgreiche Umsetzung umweltpolitischer Ziele und die ökonomische Theorie des Föderalismus in einem engen Zusammenhang.

Für eine nachhaltige Regionalentwicklung ist die Art und Intensität der Landnutzung von herausragender Bedeutung, die wiederum davon abhängig ist, inwieweit Ressourcen-, Natur-, Landschafts- und Umweltschutz im regionalen Kontext verwirklicht sind. Aufgrund des in Deutschland verfassungsrechtlich garantierten Selbstverwaltungsrechtes von Gemeinden und Gemeindeverbänden sowie der damit verbundenen Planungshoheit bezüglich der Flächennutzung wird eine nachhaltige Landnutzung maßgeblich auf kommunaler Ebene bestimmt. Eine wesentliche Frage besteht also darin, wie ökologisch notwendige und wünschenswerte Landnutzungspraktiken als Beitrag kommunaler Gebietskörperschaften zu einer nachhaltigen Regionalentwicklung gefördert werden können – und ob es adäquate fiskalische Instrumente gibt, die eine nachhaltige Landnutzung unterstützen können. In diesem Zusammenhang wird seit einiger Zeit der kommunale Finanzausgleich verstärkt diskutiert (SRU 1996; Ewers et al. 1997; Rose 1999). Der folgende Beitrag widmet sich diesem fiskalischen Instrument, indem das Konzept nachhaltiger Entwicklung als normative Leitidee für die Ausgestaltung der Finanzbeziehungen im föderativen Staat herangezogen wird.

3.2 Die kommunale Ebene und ihre Einnahmequellen

Die kommunale Ebene – dazu gehören kreisfreie Städte, Landkreise und Bezirksverbände sowie kreisangehörige Gemeinden – zeichnet sich in der Bundesrepublik Deutschland durch eine relativ hohe Autonomie aus, die durch das verfassungsrechtlich garantierte kommunale Selbstverwaltungsrecht begründet ist. Den Kommunen kommt je nach Rechtsgebiet eine gewisse Eigenständigkeit in der Umsetzung von Landes- und Bundesrecht zu. Im Bereich der Bauleit- und Flächennutzungsplanung besitzen sie die Planungshoheit, was für Fragen der Landnutzung von besonderer Relevanz ist. Zur Durchführung ihrer in eigener Verantwortung zu regelnden Angelegenheiten der örtlichen Gemeinschaft sowie zur Erledigung der ihnen von zentraler Ebene übertragenen Aufgaben benötigen kommunale Gebietskörperschaften besondere fiskalische Instrumente (Zimmermann 1999; Birke 2000). Hierzu zählen eigene Einkommensquellen wie die kommunalen Steuern, Abgaben und Gebühren. Darüber hinaus wird ein beträchtlicher Teil der kommunalen Einnahmen über die Landeszuweisungen bereitgestellt, da die bundesdeutsche Finanzverfassung von einer (zwar modifizierten) Zweistufigkeit ausgeht und die Bundesländer für eine ausreichende Finanzausstattung der Gemeinden und Gemeindeverbände Sorge tragen müssen (vgl. Art. 106, VII und IX, Art. 107 II i.V.m. Art. 28 II GG sowie Lenk 2001, S. 53f.). Näheres regeln die kommunalen Finanzausgleichsgesetze der Bundesländer. Der Anteil der Landeszuweisungen am kommunalen Budget in den alten Bundesländern beträgt ungefähr 30 %. Die Haushalte der Kommunen in den neuen Bundesländern hängen sogar mit rund 50 % von den Mitteln aus dem kommunalen Finanzausgleich ab. Erst mit diesen Mitteln ist es den Kommunen möglich, ihre unterschiedlichen öffentlichen Aufgaben zu erfüllen. Abbildung 3.1 vermittelt einen Überblick über die Herkunft, den Freiheitsgrad und die Verwendung der Mittel aus dem kommunalen Finanzausgleich für die alten Bundesländer.

In einer Reihe deutscher Bundesländer wird der überwiegende Anteil der vertikalen Transfers in Form von allgemeinen Zuweisungen an kommunale Gebietskörperschaften gegeben, die zur freien Verwendung zur Verfügung stehen. Die Schlüsselzuweisungen als zentraler Bestandteil des kommunalen Finanzausgleiches bestimmen sich neben der Finanzkraft der jeweiligen Gebietskörperschaft durch ihren Finanzbedarf. Die Finanzkraft ergibt sich durch die Berücksichtigung der gemeindlichen Steuereinnahmen, d.h. des Einkommenssteueranteils, der Gewerbesteuer unter Berücksichtigung der Gewerbesteuerumlage und der Grundsteuer.

Der Finanzbedarf wird in der Regel über einen Gesamtansatz bestimmt, der sich aus der Summe von Hauptansatz und Neben- bzw. Leistungsansätzen ergibt.

Mittelherkunft	**Freiheitsgrad**	**Verwendung**
Steuerverbund obligatorisch – Einkommensteuer – Körperschaftsteuer – Umsatzsteuer fakultativ – Länderfinanzausgleich – Gewerbesteuerumlage – Grunderwerbsteuer – Vermögensteuer – andere Landessteuern 70 %	Allgemeine Zuweisungen disponibel 56 %	Schlüsselzuweisungen 44 % Bedarfszuweisungen 1 % Investitionspauschale u.a. allgemeine Zuweisungen 11 %
Sonstige Landes- und Bundesmittel 30 %	Spezielle Zuweisungen zweckgebunden Investive und laufende Zweckzuweisungen/ Erstattungen 44 %	Schule, Kultur 7 % Soziales, Gesundheit 21 % Öffentliche Einrichtungen, Wirtschaftliche Unternehmen 6 % Bauwesen, Verkehr u.a. 10 %

Nach den Finanzausgleichsgesetzen der alten Länder 1997.
Quelle: Karrenberg und Münstermann (2000, S. 55)

Abbildung 3.1: Struktur des kommunalen Finanzausgleichs (alte Bundesländer).

Figure 3.1: Structure of fiscal equalisation at the local level (old German *Länder*).

Der Hauptansatz basiert auf dem Indikator Einwohnerzahl der betroffenen Gebietskörperschaft. In der überwiegenden Zahl der Bundesländer wird die Einwohnerzahl noch zusätzlich gewichtet, zum Teil progressiv mit steigender Einwohnerzahl. Dementsprechend ergibt sich, dass mit zunehmender Einwohnerzahl einer Gebietskörperschaft die Höhe der Schlüsselzuweisungen mitunter progressiv steigt. Dieser Zusammenhang, der sich auf Brecht (1932) und Popitz (1932) zurückführen lässt[1], wird heute in Theorie und Praxis kontrovers diskutiert. In

[1] Brecht (1932, S. 6) hat das „Gesetz von der progressiven Parallelität von öffentlichen Ausgaben und Einwohnerzahl“ formuliert. Popitz (1932, S. 130) argumentierte, dass Stadtbewohner eine höhere Nachfrage nach öffentlichen Gütern besitzen als die ländliche Bevölkerung.

der Folge haben die Bundesländer Rheinland-Pfalz und Schleswig-Holstein die Einwohnergewichtung bereits abgeschafft.

Ergänzend zum einwohnerbasierten Hauptansatz existieren in den kommunalen Finanzausgleichsgesetzen der Bundesländer Nebenansätze zur Verteilung der Schlüsselzuweisungen, die z.B. die Anzahl der Schüler, Soziallasten oder zentralörtliche Funktionen der jeweiligen Gebietskörperschaften berücksichtigen. Während der Hauptansatz den allgemeinen und durchschnittlichen Finanzbedarf einer Kommune zu berücksichtigen hat, sollten die Nebenansätze gemeindespezifische, regionale und überregionale finanzielle Sonderlasten erfassen.

Neben den allgemeinen Zuweisungen gibt es noch die speziellen, zweckgebundenen Zuweisungen an die Kommunen. Sie speisen sich aus der Finanzausgleichsmasse, werden aber darüber hinaus durch weitere Bundes- und Landesmittel ergänzt. Mit den Zweckzuweisungen werden in der Regel Lenkungsabsichten verfolgt und die Ausgaben der Kommunen an den landespolitischen Zielen orientiert. Die Mittelzuweisung erfolgt mitunter nur auf Antragstellung, für ihre Bewilligung wird häufig eine Eigenbeteiligung der Kommune an den zu finanzierenden Projekten vorausgesetzt.

3.3 Bedeutung und Rechtsverbindlichkeit ökologischer Aufgaben

Für die vorliegende Untersuchung werden die öffentlichen Aufgaben einer Gebietskörperschaft entsprechend dem Leitbild einer nachhaltigen Entwicklung in soziale, ökonomische und ökologische Aufgaben unterteilt. Aus der Sicht der ökonomischen Theorie des Föderalismus gilt es, die zweckmäßige Zuordnung öffentlicher Aufgaben auf verschiedene föderale Ebenen vorzunehmen, wobei in der Regel eine Differenzierung von Einnahmen- und Ausgabenmaßnahmen nach allokativen, distributiven und stabilisierenden Funktionen stattfindet (vgl. stellvertretend Musgrave 1959, Kapitel 1). Sowohl für die distributiven Funktionen (z.B. Einkommensverteilung), die hier vereinfachend mit den sozialen Aufgaben gleichgesetzt werden, als auch für die makroökonomischen Stabilisierungsfunktionen gilt, dass hierfür in der Regel zentrale Gebietskörperschaften verantwortlich sein sollten. Bezüglich der Allokationsfunktion gilt das Prinzip der fiskalischen Dezentralisierung. Die Bereitstellung von zahlreichen öffentlichen Gütern und Dienstleistungen lässt sich – bei Abwesenheit von Skalen- und externen Effekten – am effizientesten gestalten, wenn Produktion und Konsum der niedrigst möglichen Gebietskörperschaft zugewiesen werden (Musgrave 1959; Oates 1972). Im Folgenden wird zu hinterfragen sein, inwieweit im Rahmen der allokativen Funktion ökologische öffentliche Aufgaben im kommunalen Finanzausgleich berücksichtigt werden. Dazu werden die allokativen Aufgaben in ökologische und nicht-ökologische Aufgaben unterteilt[2].

Wie lassen sich ökologische öffentliche Aufgaben definieren? In erster Näherung sind darunter Aufgaben des Natur- und Umweltschutzes zu verstehen. Die Notwendigkeit staatlichen Handelns bei der Bereitstellung von Natur- und Umweltschutz ist erforderlich, da viele dieser

[2] Für die vorliegende Analyse genügt die Unterscheidung von ökologischen und nicht-ökologischen Aufgaben, wobei letztere sowohl soziale als auch ökonomische Aufgaben umfassen.

Leistungen durch das Kriterium der Nichtausschließbarkeit gekennzeichnet und den öffentlichen Gütern zuzurechnen sind. Öffentliche Aufgaben des Natur- und Umweltschutzes beziehen sich einerseits auf den Schutz und die nachhaltige Nutzung von natürlichen Ressourcen, von Lebewesen, Ökosystemen und Landschaften. Nach den §§ 1 und 2 des Bundesnaturschutzgesetzes sind mit Naturschutz und Landschaftspflege die Lebensgrundlagen des Menschen, auch als Vorraussetzung für seine Erholung in Natur und Landschaft, nachhaltig zu sichern. Insofern werden im weiteren Verlauf der Untersuchung öffentliche Aufgaben im Zusammenhang mit den Bereichen Boden, Wasser, Naturschutz und Landschaftspflege, aber auch der Erholung betrachtet. Andererseits beziehen sich entsprechende öffentliche Aufgaben auf die Regulierung von Umweltbelastungen, d.h. von negativen Auswirkungen menschlicher Aktivitäten auf die sie umgebende Umwelt in Form von beispielsweise Emissionen, Abfällen, Altlasten oder Landschaftszerstörung. In konkreter Form ließen sich hier die Abwasser- und Abfallentsorgung, aber auch die Altlastensanierung und Landschaftsrekultivierung nennen.

Um diesem weiten Verständnis öffentlicher Aufgaben im Rahmen von Naturschutz- und Umweltpolitik gerecht zu werden, aber auch in Anlehnung an die drei Dimensionen der Nachhaltigkeit, wird der Begriff „ökologisch“ im Zusammenhang mit den genannten öffentlichen Aufgaben verwendet. Manche dieser Aufgaben, wie z.B. die Erholung oder Landschaftsrekultivierung infolge von Bergschäden, lassen sich nicht eindeutig den ökologischen Aufgaben zuordnen, denn sie besitzen gleichermaßen eine soziale bzw. ökonomische Komponente. Andere Aufgaben wiederum haben bereits eine lange Tradition im Katalog kommunaler öffentlicher Aufgaben, da sie wie die Abwasserentsorgung vor allem aus dem Blickwinkel kommunaler Daseinsvorsorge bzw. stadthygienischer Erwägungen von Bedeutung sind. Bei einer vollständigen Aufzählung ökologischer Aufgaben dürfen diese Aufgaben dennoch nicht fehlen, da sie sich auch durch ökologisch relevante Komponenten auszeichnen. Ökologische Aufgaben gilt es auf allen administrativen Ebenen in föderalen Staatengebilden zu berücksichtigen und angemessen zu finanzieren. Diese Forderung baut im Wesentlichen auf zwei Argumenten auf:

Das erste Argument basiert auf der Entwicklung von Naturschutz- und Umweltpolitik und den entsprechenden rechtsverbindlichen Regelungen. Seit den 1970er Jahren ist ein umfangreiches Gesetzeswerk geschaffen worden, das den verschiedenen administrativen Ebenen in der Bundesrepublik Deutschland zahlreiche ökologische Aufgaben zugewiesen hat. In Abhängigkeit von Ausgestaltung und Wechselbeziehungen zwischen europa-, bundes- und landesweiten, regionalen und kommunalen ökologischen öffentlichen Aufgaben entstehen entsprechende Ausgaben. Im Gegensatz zu den wesentlich länger bestehenden und etablierten öffentlichen Aufgaben im sozioökonomischen Bereich, die mit vergleichsweise höheren finanziellen Ressourcen zur Bereitstellung der betreffenden Güter und Leistungen ausgestattet sind, wird für den Naturschutz- und Umweltbereich vielfach eine Unterfinanzierung beklagt (vgl. z.B. Kaule 1991, S. 353ff.; Henle 1995; Steffens 1997; Hessischer Landtag 1997)[3]. Da

[3] Auch die Vollzugsdefizite in der Umweltpolitik können zum Teil der Unterfinanzierung dieses Aufgabenfeldes geschuldet sein.

der kommunale Finanzausgleich u.a. der Sicherung der finanziellen Ressourcen zur Erledigung kommunaler öffentlicher Aufgaben dient, ist dieser im Hinblick auf die systematische Integration ökologischer Aufgaben und Indikatoren zu untersuchen.

Das zweite Argument steht in Verbindung mit der Mitgliedschaft der Bundesrepublik Deutschland in der Europäischen Union und damit der rechtlichen Verbindlichkeit der europäischen Verträge, Richtlinien und Verordnungen. Seit dem Amsterdamer Vertrag von 1997 ist das Leitbild der nachhaltigen Entwicklung Bestandteil des europäischen Vertragswerkes.[4] Gemäß der Präambel des Vertrages von Amsterdam soll der wirtschaftliche und soziale Fortschritt der europäischen Völker „unter Berücksichtigung des Grundsatzes der nachhaltigen Entwicklung“ verfolgt werden. Darüber hinaus wird nachhaltige Entwicklung im neuen Artikel 2 des EU-Vertrages auf die gleiche Ebene mit den Zielen der Förderung des wirtschaftlichen und sozialen Fortschritts sowie eines hohen Beschäftigungsniveaus gestellt (Frenz und Unnerstall 1999, S. 173f.). Das Leitbild einer nachhaltigen Entwicklung hat sich somit zu einem führenden Konzept für Politikformulierung und -gestaltung in den Mitgliedstaaten der Europäischen Union entwickelt, das in Verbindung mit dem Subsidiaritätsprinzip auf allen relevanten administrativen Ebenen umzusetzen ist.

3.4 Ökologische Aufgaben in den kommunalen Finanzausgleichsgesetzen

Ökologische öffentliche Aufgaben werden in den kommunalen Finanzausgleichsgesetzen der Bundesländer[5] bereits in den verschiedensten Formen berücksichtigt[6]. Sie finden Erwähnung im Rahmen von Nebenansätzen, pauschalen Zuweisungen, besonderen Bedarfszuweisungen, Zweckzuweisungen, Darlehen, Zinshilfen und Zuschüssen, um nur eine Auswahl zu nennen. Teilweise finden Vorwegentnahmen aus der Gesamtschlüsselmasse statt, teilweise werden Zuweisungen benannt, deren Finanzierung außerhalb der Verbundmasse geregelt wird. Mitunter gibt es auch Schlüssel- oder Zweckzuweisungen für ökologische Aufgaben, obwohl dies

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4 Dies betrifft sowohl den Vertrag über die Europäische Union (EUV) als auch den Vertrag zur Gründung der Europäischen Gemeinschaft (EGV).

5 Die Stadtstaaten Berlin, Bremen und Hamburg werden von der Analyse ausgenommen. Aufgrund ihrer Sonderstellung sind hier Landes- und kommunale Aufgaben schwer zu trennen; kommunale Finanzausgleichsgesetze existieren nicht.

6 Die nun folgenden Aussagen basieren, soweit nicht durch andere Quellen belegt, auf einer Analyse der kommunalen Finanzausgleichsgesetze der deutschen Bundesländer: Baden-Württemberg: FAG i.d.F. v. 1.1.2000. GBl. 2000, S. 14. - Bayern: FAG i.d.F.d.Bek. v. 23.2.2000. GVBl. 2000, S. 70, zul. geänd. d. G. v. 22.12.2000. GVBl. 2000, S. 940. - Brandenburg: GFG 2000 v. 15.2.2000. GVBl. 2000, S. 2. - Hessen: FAG i.d.F. v. 18.3.1997. GVBl. 1997, S. 58, zul. geänd. d. G. v. 19.12.2000. GVBl. 2000, S. 553. - Mecklenburg-Vorpommern: FAG i.d.Bek. v. 12.1.2000. GVOBl. 2000, S. 2, geänd. d. G. v. 18.12.2000. GVOBl. 2000, S. 559. - Niedersachsen: NFAG i.d.F.d.Bek. v. 26.5.1999. GVBl. 1999, S. 116, geänd. d. G. v. 22.6.2000. GVBl. 2000, S. 138. - Nordrhein-Westfalen: GFG 2001 v. 3.4.2001. GV NRW 2001, S. 172. - Rheinland-Pfalz: LFAG v. 30.11.1999. GVBl. 1999, S. 415. - Saarland: KFAG v. 12.7.1983. Abl. 1983, S. 462, zul. geänd. d. G. v. 7.6.2000. Abl. 2000, S. 1021. - Sachsen: FAG v. 19.12.2000. SächsGVBl. 2001, S. 2. - Sachsen-Anhalt: FAG v. 1.7.1999. GVBl. 1999, S. 204. - Schleswig-Holstein: FAG i.d.F. v. 4.2.1999. GVOBl. 1999, S. 47, zul. geänd. d. G. v. 28.12.2000. GVOBl. 2000, S. 632. - Thüringen: ThürFAG i.d.F.d.Bek. v. 9.2.1998. GVBl. 1998, S. 15, zul. geänd. d. G. v. 19.12.2000. GVBl. 2000, S. 381.

aus den Finanzausgleichsgesetzen nicht wörtlich zu entnehmen ist. So gewährt Baden-Württemberg Zuweisungen für die Altlastensanierung, Wasserversorgung und Abwasserentsorgung im Rahmen der Kommunalen Investitionspauschale und des Kommunalen Investitionsfonds (Finanzministerium Baden-Württemberg 2000, S. 18). Des Weiteren ist zu berücksichtigen, dass es in allen Bundesländern, zum Teil auch auf Bundes- und EU-Ebene, zweckgebundene Zuwendungen für ökologische Aufgaben außerhalb der Regelungen in den kommunalen Finanzausgleichsgesetzen gibt, die im Rahmen einer Vielzahl von umwelt- und naturschutzbezogenen Förderrichtlinien geregelt sind. Für eine vollständige Analyse der Zuweisungen für ökologische Aufgaben wären neben den Finanzausgleichsgesetzen auch alle weiteren Regelungen einzubeziehen, in deren Rahmen kommunale Gebietskörperschaften antragsberechtigt sind. Obwohl dies nicht Ziel der vorliegenden Analyse ist, gilt es, diese Aspekte im Auge zu behalten, wenn im Folgenden der Stand der Berücksichtigung ökologischer Aufgaben in den kommunalen Finanzausgleichsgesetzen der Flächenländer präsentiert wird. Trotz der erwähnten Einschränkungen macht es Sinn, die Finanzausgleichsgesetze einer gründlichen Analyse zu unterziehen. Die Zuweisungen aus dem kommunalen Finanzausgleich stellen eine wesentliche kommunale Einnahmequelle dar. Dies betrifft sowohl die allgemeinen Zuweisungen ohne Verwendungsauflage, die in vielen Bundesländern den Hauptanteil ausmachen, als auch die speziellen, zweckgebundenen Zuweisungen. Insofern zielt die Analyse der kommunalen Finanzausgleichsgesetze hinsichtlich der Berücksichtigung ökologischer Aufgaben zwar einerseits auf die konkreten ökologischen Aufgabenfelder, die im Rahmen der Finanzausgleichsgesetze gefördert werden. Andererseits ist auf die gewählten Formen der Zuweisungen einschließlich der dazugehörigen Indikatoren zu achten und inwieweit diese ökologische Aspekte einschließen.

3.4.1 Der Indikator Fläche als indirekter Ansatz

Flächenbezogene Ansätze können als ein erster, indirekter Schritt der Anerkennung ökologischer Aufgaben in den Finanzausgleichsgesetzen betrachtet werden. Dies liegt in der Bedeutung von Fläche und flächenintensiven Landnutzungen für zahlreiche ökologische Aufgaben begründet. Hierunter fallen Aufgaben des Ressourcenschutzes wie z.B. die Ausweisung von Trinkwasser-, Natur- oder Landschaftsschutzgebieten, aber ebenso Aufgaben des Umweltschutzes wie etwa die Abwasserentsorgung[7].

Die Berücksichtigung eines Flächenindikators für die Allokation von Zuweisungen ist besonders für großflächige Gemeinden und Landkreise relevant. Je flächenintensiver eine Gemeinde oder ein Landkreis und je weiter die Entfernung von verdichteten Räumen oder Städten, desto höher ist die Bedeutung flächenbezogener Indikatoren. Dies betrifft vor allem die ländlichen und naturnäheren Räume mit geringer Einwohnerdichte, die sich hinsichtlich ihrer Landnutzung durch einen größeren Anteil land- und forstwirtschaftlicher Nutz- sowie naturschutzfachlich wertvoller Flächen auszeichnen.

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[7] Für eine ausführliche Analyse der zunehmenden Kosten der Abwasserentsorgung mit abnehmender Siedlungsdichte vgl. Seitz (2000).

Aus den gegebenen Zusammenhängen heraus überrascht es nicht, dass die Berücksichtigung flächenbezogener Indikatoren seit langem vom Deutschen Landkreistag gefordert wird (Henneke 2001)[8]. Dies liegt in der Kostenwirksamkeit des Faktors Fläche für die Finanzierung bestimmter kommunaler Aufgaben begründet. Je flächenintensiver eine Gemeinde oder ein Landkreis in Verbindung mit einer geringen Einwohnerdichte ist, desto teurer gestaltet sich die Bereitstellung bestimmter öffentlicher Güter und Leistungen. So sind zusätzliche Kosten in Verbindung mit ökologischen, sozialen und ökonomischen öffentlichen Aufgaben zu erwarten wie z.B. dem Naturschutz, der Wasserversorgung, der Abwasser- und Abfallentsorgung, dem Landwirtschaftswesen, dem öffentlichen Nahverkehr sowie dem Schul- und Gesundheitswesen (Henneke 2001; Bergmann 1999; Bizer et al. 1998, S. 51).

Deshalb berücksichtigt auch die Mehrzahl der Finanzausgleichsgesetze der deutschen Bundesländer flächenbezogene Indikatoren, lediglich Niedersachsen bildet hier eine Ausnahme. In Brandenburg, Mecklenburg-Vorpommern, Rheinland-Pfalz und Sachsen-Anhalt wird der Indikator Fläche zur Berechnung der Bedarfsmesszahl der Gebietskörperschaft herangezogen, die sich unmittelbar auf die Höhe der zu beziehenden Schlüsselzuweisungen auswirkt (Henneke 2001)[9]. In manchen Fällen findet der Flächenindikator Eingang in die Berechnung von Zuweisungen, die der pauschalen Förderung investiver Maßnahmen dienen[10]. Die meisten Bundesländer wählen – zum Teil zusätzlich, meist aber ausschließlich – den Weg, diejenigen Aufgabenfelder direkt zu benennen, die mit zunehmender Fläche teurer werden. Dies geschieht in der Regel durch die anteilige Berücksichtigung des Flächenindikators bei der Bestimmung von Zweckzuweisungen, die sich auf Aufgabenfelder wie beispielsweise Straßenbau und -instandhaltung, öffentlicher Nahverkehr, Schülerbeförderung oder Abwasserentsorgung beziehen. Obgleich aus theoretischer Sicht auch ökologische Aufgaben mit zunehmender Fläche zu einer Kostenerhöhung führen können, dominieren bei der praktischen Ausgestaltung der Zweckzuweisungen die sozialen und ökonomischen Aufgabenfelder das Bild.

3.4.2 Die direkte Berücksichtigung ökologischer Aufgaben

Neben dem indirekten Indikator Fläche werden ökologische öffentliche Aufgaben in den Finanzausgleichsgesetzen auch in direkter Form benannt, dabei bedienen sich die Bundesländer unterschiedlicher Wege. Lediglich Sachsen-Anhalt und Niedersachsen bilden hier eine Aus-

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[8] Hier erfahren flächenorientierte Ansätze eine besondere Wertschätzung in der jährlichen Analyse des kommunalen Finanzausgleiches unter besonderer Berücksichtigung der Landkreise.

[9] In der Regel wird der Indikator Fläche im Nebenansatz zur Berechnung der Bedarfsmesszahl herangezogen. Sachsen-Anhalt berücksichtigt ihn als einziges Bundesland im Hauptansatz, wobei hier seit 1999 beabsichtigt ist, den Flächenfaktor bis 2002 zurückzuführen.

[10] So in Nordrhein-Westfalen: § 17 (2) Gemeindefinanzierungsgesetz (GFG) nennt pauschale Zuweisungen für kommunale Investitionsmaßnahmen, die zu 5/6 nach der Einwohnerzahl und zu 1/6 nach der Gebietsfläche verteilt werden. Die pauschale Förderung investiver Maßnahmen im Abwasserbereich nach § 17 (4) GFG gewichtet die Gebietsfläche sogar mit 2/3, während die Einwohnerzahl mit 1/3 in die Verteilung eingeht.

nahme: Diese beiden Bundesländer benennen in ihren Finanzausgleichsgesetzen überhaupt keine ökologischen Aufgaben in direkter Form[11].

Was die Formen der Berücksichtigung ökologischer Aufgaben angeht, reservieren einige Bundesländer Mittel im Rahmen von Vorwegentnahmen aus der Finanzausgleichsmasse. So entnimmt Bayern vorab Mittel für Verstärkungsbeträge für Zuwendungen zum Bau von Abfallentsorgungsanlagen (Bayerisches Finanzausgleichsgesetz Art. 1 (2) und Art. 10c). In Baden-Württemberg werden der Finanzausgleichsmasse 50 % des Erstattungsbetrages für die Erkundung oder Sanierung von Bodenbelastungen entnommen, falls die Kosten von den unteren Bodenschutzbehörden zu tragen sind (Finanzausgleichsgesetz Baden-Württemberg § 2 Nr. 9 sowie § 10 (4) Bodenschutzgesetz Baden-Württemberg). In Sachsen bestehen Vorwegentnahmen für besondere Bedarfszuweisungen, die auch zur Unterstützung von Haushaltskonsolidierungen bei Zweckverbänden der Wasserver- und Abwasserentsorgung verwendet werden können (Sächsisches Finanzausgleichsgesetz § 3 (1) Nr. 1.b) und § 22 Nr. 7). In Mecklenburg-Vorpommern fließen über eine Vorwegentnahme 20 Millionen DM in einen Kommunalen Aufbaufonds. Aus diesem können Zinshilfen und Darlehen, in Ausnahmefällen auch Zuschüsse für den Bau von Abfallentsorgungsanlagen, den Ausbau der Wasserversorgung und Abwasserbeseitigung oder die Förderung von Naherholungs-, Landschafts- und Kulturpflegemaßnahmen gewährt werden (Finanzausgleichsgesetz Mecklenburg-Vorpommern § 6 (1) Nr. 8 und § 16).

Die Berücksichtigung ökologischer Aufgaben im Rahmen von Nebenansätzen erfolgt lediglich in Ausnahmefällen. So gibt es im Saarland einen Ergänzungsansatz für Bergschäden. Hessen, Rheinland-Pfalz und das Saarland nutzen Ergänzungsansätze in Verbindung mit Kur- und Erholungsorten.

Die weitaus zahlreichsten direkten Nennungen erfahren ökologische Aufgaben im Rahmen der Zweckzuweisungen. Mit Ausnahme der bereits erwähnten Bundesländer Niedersachsen und Sachsen-Anhalt sind diese in den Finanzausgleichsgesetzen aller Flächenländer zu finden. Tabelle 3.1 gibt einen Überblick über die Förderbereiche, die geförderten Maßnahmen und die jeweiligen Bundesländer, in deren Finanzausgleichsgesetzen entsprechende Maßnahmen genannt werden.

Die Abwasserentsorgung wird mit Abstand am häufigsten genannt (10 Länder), was nicht wundert, gehört sie doch auf kommunaler Ebene zu den zentralen und aufwendigsten öffentlichen Aufgabenbereichen (vgl. Seitz 2000). Danach folgt mit geringem Abstand der Bereich der Wasserversorgung (8 Länder). Auch die Förderung von Abfallentsorgungsanlagen (6 Länder) sowie die Erkundung und Sanierung von Altlasten (5 Länder) werden noch relativ häufig im Rahmen der Zweckzuweisungen genannt.

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[11] Allerdings erwähnt Niedersachsen in seinem Finanzausgleichsgesetz als einziges Bundesland generell keine konkreten Aufgabenbereiche. Insofern gibt das niedersächsische Finanzausgleichsgesetz keine Anhaltspunkte für eine ausreichende oder mangelnde Berücksichtigung ökologischer Aufgaben. Sachsen-Anhalt nennt dagegen zahlreiche Zuweisungen für konkrete nicht-ökologische Aufgaben, so dass hier das Fehlen ökologischer Aufgaben besonders auffällt.

Tabelle 3.1: Zweckzuweisungen für ökologische Aufgaben im Rahmen der kommunalen Finanzausgleichsgesetze.

Table 3.1: Specific-purpose transfers for ecological public functions in fiscal equalisation laws at the local level.

Förderbereiche	Geförderte Maßnahmen	Bundesländer
Boden	Erkundung und Sanierung von Altlasten, Rekultivierung	BY, BW, HE, NW, TH
Wasser	Gewässerschutz	HE
	Wasserversorgung	BY, BB, HE, MV, RP, SL, SN, TH
	Abwasserentsorgung	BY, BB, BW, HE, MV, NW, RP, SL, SN, TH
Naturschutz, Landschaftspflege und Erholung	Naturschutz und Landschaftspflege	BB, HE, MV, NW
	Kur- und Erholungsorte	BW, NW, RP
	Naherholungs- und Fremdenverkehrsmaßnahmen	BB, MV, RP, SH
Abfall	Abfallentsorgungsanlagen	BY, HE, MV, RP, SL, TH
Energie	Energiesparmaßnahmen	HE

BW, Baden-Württemberg; BY, Bayern; BB, Brandenburg; HE, Hessen; MV, Mecklenburg-Vorpommern; NW, Nordrhein-Westfalen; RP, Rheinland-Pfalz; SL, Saarland; SH, Schleswig-Holstein; SN, Sachsen; TH, Thüringen.

In sehr viel geringerem Maße werden Zweckzuweisungen für Maßnahmen des Natur- und Ressourcenschutzes gewährt. Hier sind die Bundesländer Hessen und Nordrhein-Westfalen hervorhebenswert. So unterstützt Hessen Projekte der Biotopsicherung und -vernetzung, Gewässerschutzmaßnahmen und den naturnahen Ausbau von Fließgewässern. Nordrhein-Westfalen gewährt Zweckzuweisungen für die Bodendenkmalspflege, den Erwerb und die Nutzbarmachung von Brachflächen sowie für die ökologische Gestaltung des Emscher-Lippe-Raums.

In Zusammenhang mit Maßnahmen des Natur- und Ressourcenschutzes, der Erholung sowie einiger anderer ökologischer Aufgaben muss an dieser Stelle auch noch Schleswig-Holstein erwähnt werden. Hier werden für zahlreiche ökologische Aufgaben Darlehen vergeben[12]. Die Vergabe von Darlehen für ökologische Aufgaben steht dabei aber in einem gewissen Gegensatz zur Vergabe von realen Zuweisungen für zahlreiche nicht-ökologische Aufgaben.

Einen Sonderfall stellt die Förderung von Erholung dar. Fasst man die verschiedenen Formen der Gewährung von Zuweisungen für diesen Bereich zusammen, die von Nebenansätzen über pauschale Zuweisungen zu Zweckzuweisungen und Darlehen reichen, so nennen immerhin 8 Länder diesen Aufgabenbereich in ihren Finanzausgleichsgesetzen. Dies lässt sich inso-

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[12] Im Rahmen des Kommunalen Investitionsfonds nach § 19 Finanzausgleichsgesetz Schleswig-Holstein.

fern erklären, als der Bereich Erholung eine Schnittstelle zwischen den ökologischen und sozioökonomischen Aufgaben darstellt, da die entsprechenden Maßnahmen unmittelbar dem Menschen selbst zu gute kommen.

3.5 Voraussetzungen zur systematischen Integration ökologischer Aufgaben

3.5.1 Erweiterungsbedarf nach Aufgabenbereichen

Insgesamt werden also nur in Hessen und Nordrhein-Westfalen ökologische Aufgaben im umfassenderen Sinn berücksichtigt, d.h. bezogen auf vorsorgende Ressourcenschutz- und nachsorgende Umweltschutzmaßnahmen. Ansonsten besteht die Tendenz, in den Finanzausgleichsgesetzen nachsorgende Maßnahmen bzw. „End-of-pipe"-Infrastruktur wie die Abwasser- und Abfallentsorgung sowie die Sanierung von Altlasten zu nennen. Mit Ausnahme des Bereiches Erholung und der (Trink-)Wasserversorgung als infrastruktureller Aufgabe werden Ressourcen- und Naturschutz nur selten erwähnt (vgl. Abbildung 3.2).

Status quo	Erweiterungsbedarf
Wasserver- und Abwasserentsorgung	Naturschutz und Landschaftspflege
Abfallentsorgung	Gewässerschutz
Altlastensanierung	Bodenschutz
⇩	⇩
Dominanz nachsorgender und infrastruktureller Aufgaben	Verstärkung vorsorgender und intergenerativer Aufgaben

Abbildung 3.2: Öffentliche Aufgaben des Natur- und Umweltschutzes im kommunalen Finanzausgleich.

Figure 3.2: Public functions of nature conservation and environmental protection in fiscal equalisation at the local level.

Dieses Ergebnis steht in enger Verbindung mit der historischen Entwicklung kommunaler öffentlicher Leistungen. So gehören die Wasserversorgung sowie die Abwasser- und Abfallentsorgung mit ihren infrastrukturellen Einrichtungen seit langem zu den Kernbereichen der kommunalen Daseinsvorsorge, die ihrem Wesen nach der kommunalen Selbstverwaltung unterstehen. Diese Aufgabenbereiche zeichnen sich dadurch aus, dass sie mit den (Grund-)Bedürfnissen der Einwohner einer Gebietskörperschaft verknüpft sind (Gewährleistung einer Grundversorgung sowie Notwendigkeit hygienischer Mindeststandards). Aufgrund ihrer unmittelbaren Umweltrelevanz werden sie jedoch entsprechend der weiter oben entwickelten Systematik auch den ökologischen Aufgaben zugeordnet. Die Finanzierung dieser Aufgaben erfolgt grundsätzlich über Gebühren und Beiträge, die für die Inanspruchnahme der entsprechenden öffentlichen Leistungen zu zahlen sind, der kommunale Finanzausgleich dient in der Regel lediglich dem Abfangen von Spitzenlasten und Sonderbedarfen. Deshalb werden diese Bereiche bei der Diskussion um die Ökologisierung des kommunalen Finanzausgleiches eine vergleichsweise geringe Rolle spielen.

Dagegen gehören Naturschutz und Landschaftspflege sowie der Gewässer- und Bodenschutz im Sinne des abiotischen und biotischen Naturschutzes zu denjenigen ökologischen öffentlichen Aufgaben, wie sie zur Gewährleistung einer bestimmten Natur- und Umweltqualität auch unabhängig von menschlichen Bedürfnissen bestehen. Eine direkte Finanzierung über Gebühren und Beiträge entfällt weitgehend aufgrund des Fehlens nutzerbezogener öffentlicher Leistungen. Dies hängt entscheidend mit der Nichtanwendbarkeit des Ausschlussprinzips bei der Bereitstellung entsprechender Leistungen zusammen. Zusätzlich befinden sich ausgedehnte, naturschutzfachlich wertvolle Gebiete häufig in Regionen, die durch geringe Einwohnerdichten und wirtschaftliche Strukturschwäche gekennzeichnet sind. Damit bewegen sich sowohl die eigenen Einnahmen der betroffenen Gebietskörperschaft als auch die Zuweisungen über den kommunalen Finanzausgleich auf vergleichsweise niedrigem Niveau. Folglich sind die Einbeziehung des Naturschutzes sowie entsprechender Indikatoren in den kommunalen Finanzausgleich notwendig, um dem finanzwissenschaftlichen Grundsatz der Konnexität Rechnung zu tragen. Danach sollten mit der Zuweisung öffentlicher Aufgaben zugleich auch die zur Erfüllung erforderlichen Ausgaben zur Verfügung gestellt werden.

Eine weitere Begründung für die notwendige Integration des Aufgabenbereiches Naturschutz basiert auf dem Prinzip der fiskalischen Äquivalenz (Olson 1969). Danach sollte eine Übereinstimmung zwischen denjenigen hergestellt werden, die einen Nutzen aus der Bereitstellung eines öffentlichen Gutes ziehen und jenen, die für die Bereitstellung dieses Gutes zahlen. So werden auf überkommunaler Ebene je nach naturräumlichen Gegebenheiten (z.B. Seltenheit vorkommender Tier- und Pflanzenarten) bestimmte Flächen als Vorrang- oder Schutzgebiete (zum Teil mit internationalem Rang) ausgewiesen, was auch gegen die Präferenzen der Einwohner der betroffenen Kommunen stattfinden kann. Die positiven räumlichen externen Effekte („spillover“) können weit über die kommunale Ebene hinausreichen[13], wogegen diverse Kosten und Belastungen durch Nutzungseinschränkungen überwiegend auf kommunaler Ebene verbleiben. Olson (1969) hat für diese Problematik bereits einen Lösungsansatz vorgeschlagen, obgleich er für andere öffentliche Güter wie z.B. Erziehung und Ausbildung entwickelt wurde. Solange sich bei Abwesenheit von Skaleneffekten die Bereitstellung bestimmter öffentlicher Güter und Leistungen auf kommunaler Ebene empfiehlt und das ist in Zusammenhang mit vielen Aspekten der Landnutzung der Fall, sollten räumliche Externalitäten durch Zuweisungen im Rahmen der Finanzausgleichsbeziehungen von einer zentralen an eine dezentrale Gebietskörperschaft internalisiert werden.

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[13] So etwa bei Biosphärenreservaten, FFH-Gebieten oder internationalen Vogelschutzgebieten.

3.5.2 Erweiterungsbedarf nach Zuweisungsarten

Betrachtet man zusammenfassend die Formen der Berücksichtigung ökologischer Aufgaben in den kommunalen Finanzausgleichsgesetzen, so dominieren in der Mehrzahl der Bundesländer eindeutig die Zweckzuweisungen. Zweckzuweisungen stellen nach vorherrschender finanzwissenschaftlicher Sicht die geeignete Form der Zuweisung zur Internalisierung von Spillover-Effekten dar (Fischer 1988, S. 65f.). Damit wäre diese Zuweisungsart geboten, um die Spillover-Effekte des abiotischen und biotischen Naturschutzes im Rahmen des kommunalen Finanzausgleiches zu berücksichtigen. Da die Gemeinde im Fall von räumlichen positiven externen Effekten auch einen eigenen Nutzen an der entsprechenden Leistung hat, der in der Gemeinde verbleibt, wäre dieser gemeindeinterne Nutzen im Rahmen einer Eigenbeteiligung der Gemeinde zu berücksichtigen.

Übergeordnete Gebietskörperschaften können darüber hinaus ganz allgemein die Lenkungsabsicht verfolgen, kommunale Gebietskörperschaften dazu zu bewegen, mehr Mittel für den abiotischen und biotischen Naturschutzes zu verausgaben. Hierfür wären im Rahmen des kommunalen Finanzausgleiches entweder allgemeine Investitionszuweisungen mit Verwendungsnachweis für einen entsprechend zu bestimmenden Bereich denkbar (z.B. den abiotischen und biotischen Naturschutz) oder aber zweckgebundene Zuweisungen, die für spezifische Vorhaben innerhalb dieses Bereiches zur Verfügung gestellt werden. Bei diesen Zuweisungsarten ist aber zu berücksichtigen, dass eine Nichtnennung im Rahmen der Finanzausgleichsgesetze noch nicht bedeutet, dass es für bestimmte Aufgaben keine Zuwendungen gibt. Zweckgebundene Zuwendungen für ökologische Aufgaben werden in den Ländern ausnahmslos auch über zahlreiche Förderrichtlinien geregelt. Insofern ist für den Bereich der zweckgebundenen Zuwendungen aufgrund der vorliegenden Analyse kein abschließendes Urteil möglich.

Anders ist die Lage bei den allgemeinen Zuweisungen einzuschätzen. Mit Ausnahme des Indikators Fläche[14], dem Aufgabenbereich Erholung sowie in einem Einzelfall, dem Bereich Bergschäden, findet man keine ökologischen Aufgaben in den Nebenansätzen der Finanzausgleichsgesetze, geschweige denn eine Berücksichtigung im Hauptansatz zur Berechnung der Schlüsselzuweisungen. Damit gibt es keinen Haupt- oder Nebenansatz, der auf einem Indikator basiert, der ganz allgemein ökologische Aufgaben abbildet, wie es vergleichsweise mit der Berücksichtigung von Einwohnerzahlen für sozioökonomische Aufgaben der Fall ist. Einwohnerzahlen können zwar in gewissem Umfang als Indikator für nutzerbezogene Umweltschutzaufgaben herangezogen werden, wie z.B. für die Trinkwasserversorgung sowie die Abwasser- und Abfallentsorgung, sie versagen jedoch unweigerlich bei Aufgaben, die mit dem abiotischen und biotischen Naturschutz in Verbindung stehen[15]. Sie gehen aufgrund des Fehlens entsprechender Indikatoren weder in den allgemeinen Finanzbedarf kommunaler Ge-

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[14] Dieser repräsentiert jedoch nur indirekt und partiell ökologische Aufgaben.

[15] In diesem Fall geht es um diejenigen naturschutzbezogenen öffentlichen Aufgaben, die auf kommunaler Ebene zu erledigen sind bzw. erledigt werden können und deren Nutzen in der kommunalen Gebietskörperschaft verbleibt.

bietskörperschaften ein, noch werden über die Nebenansätze spezifische Sonderlasten erfasst. Insofern sind diese Aufgaben in den gegenwärtigen Finanzausgleichsgesetzen der Bundesländer unterrepräsentiert. Dadurch kann es zu einer Unterversorgung der entsprechenden öffentlichen Güter und Leistungen kommen.

Ergänzend zu den bestehenden einwohnerbasierten Ansätzen sollte deshalb die Berechnung des kommunalen Finanzbedarfs auch auf naturschutzbezogenen Indikatoren beruhen. Hier wäre zu trennen zwischen einem Indikator, der den allgemeinen Finanzbedarf repräsentieren kann sowie Indikatoren, die gemeindespezifische, regionale und überregionale finanzielle Sonderlasten über die Nebenansätze berücksichtigen und gewisse Lenkungsfunktionen ausüben. In Analogie zum allgemein verwendeten Indikator der Einwohnerzahl für sozioökonomische Aufgaben sollten naturschutzbezogene Indikatoren ebenfalls relativ einfach zu handhaben und statistisch möglichst verfügbar sein (Bauer et al. 1996; Rose 1999)[16]. Eine noch zu leistende und nicht zu unterschätzende Forschungsaufgabe besteht schließlich in der Rechtfertigung des Zusammenhanges zwischen naturschutzbezogenen Indikatoren und der Kostenwirksamkeit bzw. Belastungsintensität der entsprechenden öffentlichen Aufgaben auf kommunaler Ebene.

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Zusammenfassung

Regionale Nachhaltigkeit bedingt die Wahrnehmung und Finanzierung ökologischer, ökonomischer und sozialer öffentlicher Aufgaben durch kommunale Gebietskörperschaften. Während die Bedeutung sozioökonomischer Aufgaben im föderalen System der Bundesrepublik Deutschland seit langem anerkannt ist, werden ökologische Aufgaben bislang nur teilweise berücksichtigt. Eine empirische Analyse der kommunalen Finanzausgleichsgesetze der 13 Flächenländer zeigt, dass nachsorgende und infrastrukturelle ökologische Aufgaben im Rahmen von Zweckzuweisungen bereits stark vertreten sind. Dagegen fehlen vorsorgeorientierte und intergenerative Aufgaben wie der Natur- und Ressourcenschutz weitgehend. Insofern ergibt sich ein Erweiterungsbedarf der kommunalen Finanzausgleichsgesetze im Hinblick auf den abiotischen und biotischen Naturschutz sowie entsprechende Indikatoren, die im Rahmen geeigneter Zuweisungsarten zu berücksichtigen wären.

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[16] Bauer et al. (1996) und Rose (1999) haben bereits eine Reihe abiotischer, biotischer und flächenbezogener Indikatoren vorgeschlagen, die in diesem Sinne auf ihre Verwendbarkeit zu prüfen wären.

Summary

Regional sustainability requires the acknowledgement and financing of ecological, economic and social public functions by local jurisdictions. Whereas fiscal federalism in Germany predominantly assigns economic and social functions to the local governmental level, ecological functions are only partially considered. An empirical analysis of the "Kommunale Finanzausgleichsgesetze" (fiscal equalisation laws) of the German states shows that end-of-the-pipe and infrastructural ecological functions are already strongly integrated by way of conditional grants. However, precautionary and intergenerational functions like nature and resource conservation are widely missing. Therefore, a comprehensive implementation of sustainable development calls for the integration of conservation issues and appropriate indicators in the system of intergovernmental fiscal relations.

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4 Compensating municipalities for protected areas. Fiscal transfers for biodiversity conservation in Saxony, Germany

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Abstract

Local conservation efforts and land-use restrictions due to protected areas are often related to benefits at higher governmental levels. This gives rise to spatial externalities that – if not adequately compensated – may lead to an underprovision of public goods and services. This article proposes two models that would expand the existing intergovernmental fiscal transfer system from the state to the local level in Saxony, Germany, to include designated protected areas; the models are based on actual administrative, social, and economic data from 2002. Conservation units (CUs) are identified in each municipality by overlaying Geographical Information System (GIS) layers of the various categories of protected areas over a total of 537 municipal borders in Saxony. The first model considers CUs within the calculation of general lump-sum transfers by “translating” CUs into the generic indicator of inhabitants. In the second model, a specified amount of the overall transfer sum is devoted to conservation according to the share of CUs a municipality holds in relation to its total area. Both approaches lead to higher transfers, especially to rural communities (though to varying degrees), thereby acknowledging the latter’s ecological services to society.

Keywords:

Biodiversity conservation, environmental federalism, fiscal transfers, Germany, local costs, protected areas, Saxony, spillover effects

4.1 Introduction

The concept of ecological tax reform was developed during the 1990s. The early focus on energy taxes was later enlarged to include fiscal policy more broadly. The debate on ecological fiscal reform, however, continues to be focused on environmental pollution, or – in economic terms – internalising negative externalities by way of taxes, charges, and fees, and thus, on greening public income. Much less has been written on internalising positive externalities, which means greening the public expenditure side. Developing fiscal transfers to compensate societal actors for the ecological goods and services they provide remains a neglected aspect of the debate. Although the German Advisory Council on the Environment has long suggested the integration of nature conservation into communal fiscal transfer systems (SRU 1996), there have been no systematic steps in this direction so far. One major argument in favour of ecological fiscal transfers relates to the spatial division of labour concerning public sector functions. Whereas the socio-economic public sector functions of urban agglomerations (such as schools, hospitals, and theatres) have long been a part of fiscal transfer schemes, functions related to ecological goods and services in rural and peripheral regions still await inclusion (Ewers et al. 1997; Ring 2002). Protected areas, for example, involve land-use restrictions that may force municipalities to forego development opportunities that would generate communal income (Figure 4.1). If protected areas were successfully included in fiscal transfers to the local level, however, their acceptance could be increased.

Figure 4.1: This highway bordering the protected landscape Leipziger Auewald exemplifies the conflict between nature protection and other forms of land use, such as transport infrastructure.

Perner and Thöne (2005) suggested ways of incorporating nature conservation into communal fiscal transfers, based mainly on indicators from spatial planning and the support of direct conservation measures. Köllner et al. (2002) developed a model based on biodiversity indicators and cantonal benchmarking for fiscal transfers to the local level in Switzerland.

This article aims to advance the debate, firstly, by describing the current state of and future prospects for ecological fiscal transfers in Germany, based on an analysis of the 13 fiscal transfer laws in the German *Länder* (federal states). Secondly, the intergovernmental fiscal transfer system from state to local level in Saxony, Germany, is presented and at the same time expanded to include a conservation indicator based on designated protected areas. Saxony was selected with a view to data availability, since the expanded transfer scheme is based on Saxon administrative, social, and economic data from 2002. Two alternative models and sets of simulation results are presented. They demonstrate for the first time in a spatially explicit way the changes in communal income that would occur if protected areas were considered in a state's fiscal transfer system.

4.2 Ecological fiscal transfers in Germany: Status quo and perspectives

Local conservation efforts as well as local land-use restrictions due to protected areas are often related to benefits at higher governmental levels. In particular, biodiversity conservation gives rise to global benefits, yet its associated costs are unequally distributed with regard to regions and economic sectors (Urfei 2002; Ring 2004; Hampicke 2005). In studies of environmental federalism, it is undisputed that the non-use values associated with natural resources in particular justify the centralisation of responsibilities (Revesz 2000; Oates 2001; List et al. 2002). In Germany, protected areas are designated by the *Länder* and at regional governmental levels, at times with and at times without the consent of local actors. Hence, land users and municipal authorities often perceive protected areas as an obstacle to development (Bauer et al. 1996), and opposition towards large protected areas can be considerable (Stoll-Kleemann 2001; Job 2008). Due to the local management and opportunity costs of providing public goods and services, spatial externalities or spillovers exist that – if not adequately compensated – lead to their underprovision (Bergmann 1999; Ring 2008a). A number of reforms are needed to reconcile the local costs and global public benefits of biodiversity conservation (Perrings and Gadgil 2003; MEA 2005). In Germany and in Europe generally, various agri-environmental programmes exist to compensate private land users for benefits foregone. For local public actors – the municipalities – only few options exist for obtaining compensation. In Saxony, one programme offers support for direct conservation measures, although the opportunity costs of protected areas imposed by spatial planning are not taken into account. Moreover, municipalities hope to generate income from designating commercial and housing development areas; this remains a major driver of land consumption.

Theoretically, fiscal federalism and the principle of fiscal equivalence provide major arguments for matching decision-making responsibilities with the costs and benefits of providing public goods and services (Buchanan 1950). Where spatial externalities exist between jurisdictions at different governmental levels, intergovernmental fiscal transfers are proposed as a suitable instrument to internalise them (Olson Jr. 1969). Fiscal transfers are an important source of local income in Germany: almost 30 percent in West Germany and more than 50 percent of average local income in eastern Germany stem from communal fiscal equalisation (Karrenberg and Münstermann 2006).

In the German *Länder*, general lump-sum transfers constitute the majority of transfers to the local level. They may be used in any way the recipient wishes, thereby acknowledging the high degree of autonomy given to the local level of government by the German Constitution. Their allocation is based on the fiscal need of a local jurisdiction in relation to its fiscal capacity (its own revenues based on local taxes). Fiscal need is determined by a "principal" approach that takes into account the general and average fiscal need. The main indicator used is the number of inhabitants, often accompanied by weighting factors linked to population size. In some states, "additional" approaches exist to take account of specific community burdens. The Free State of Saxony has an additional approach based on the number of schoolchildren; other states take account of recreational functions or the area of a municipality. The inhabitant-based indicator used for the bulk of fiscal transfers, however, generally favours urban areas as opposed to rural areas, due to the lower population densities of the latter.

Area-based indicators are a first step towards taking account of ecological public functions in fiscal transfers (Ring 2002). Henneke (2006) refers to the additional costs incurred by a large community or district area for public sector functions such as nature conservation, agricultural affairs, waste disposal, water supply, or sewage discharge. The German Association of District Councils *(Deutscher Landkreistag)* frequently requires enhanced area-related indicators in fiscal transfers. So far, only the thinly populated states of Brandenburg, Mecklenburg-Western Pomerania and Saxony-Anhalt use area as an indicator for general lump-sum transfers, along with Rhineland-Palatinate, a state with higher average population density.

North Rhine-Westphalia, Baden-Württemberg, and Hesse acknowledge ecological fiscal needs independently of a municipality's fiscal capacity. These states provide unconditional fiscal transfers relating to public sector functions in nature conservation, recreation, and environmental protection.

Table 4.1 shows fiscal transfers earmarked for ecological purposes in German fiscal equalisation laws in 2006. This is the most common method of including ecological functions in communal fiscal transfers. Infrastructure-related measures and end-of-pipe measures dominate the picture, confirming the results of a previous analysis conducted in 2001 (Ring 2002). Apart from drinking water provision, resource protection, as well as nature and biodiversity conservation are rarely supported.

Table 4.1: Earmarked ecological fiscal transfers in German fiscal equalisation laws of the German *Länder* (2006).

Area	Measures supported	*Länder* (Federal states)
Soil	Soil conservation	BW, ST
	Prospecting and remediation of contaminated sites, recultivation	BY, BW, HE, ST, TH
Water	Water protection	HE, SH, RP
	Water supply	BY, BW, HE, MV, RP, SL, SN, TH
	Sewage disposal	BY, BB, BW, HE, MV, RP, SL, SN, TH
Nature protection	Nature conservation and landscape management	HE, MV, ST
Recreation	Spas, recreation, and tourism	MV, RP, SH
Waste	Waste disposal plants	BY, HE, MV, RP, SL, TH
Energy	Energy saving measures	HE

BW, Baden-Württemberg; BY, Bavaria; BB, Brandenburg; HE, Hesse; MV, Mecklenburg-Western Pomerania; RP, Rhineland-Palatinate; SH, Schleswig-Holstein; SL, Saarland; SN, Saxony; ST, Saxony-Anhalt; TH, Thuringia.

To sum up, there is a lack of conservation-oriented public sector functions in fiscal transfers, particularly relating to the opportunity costs of protected areas. In terms of types of fiscal transfer, there is a lack of lump-sum or unconditional transfers based on conservation indicators. Such transfers would give the municipality more leeway in the use of monies received. Therefore, two alternative models are presented, each using a conservation indicator for distributing 1. lump-sum transfers and 2. unconditional ecological fiscal transfers.

4.3 Modelling fiscal transfers for protected areas in Saxony

4.3.1 The Saxon communal fiscal transfer system

The Free State of Saxony has three administrative levels: the state level, an intermediate level of administrative districts[1], and the local governmental level. The latter consists of cities that are district-independent, districts, and communities belonging to a district. In the following models, only the seven district-independent cities and 530 communities are considered (537 municipalities in total) representing Saxony's decentralised jurisdictions that make up the total state area. For the distribution of general lump-sum transfers, fiscal needs are determined by weighted inhabitants according to size classes of municipalities (principal approach) and weighted schoolchildren according to school types (additional approach). The two approaches form the "overall approach" which is then multiplied by a "base amount" to identify the fiscal need of a municipality. Different iterative calculation processes are used to determine the base amounts for district-independent cities and communities respectively. In 2002 these original

[1] *Regierungsbezirke* Chemnitz, Dresden, and Leipzig.

base amounts were 1,150.62 euros for cities and 520.81 euros for communities. Fiscal capacity is determined by a municipality's own local revenues, consisting of local land and business taxes plus local shares of income and value-added taxes (Lenk 2005). Municipalities do not receive any lump-sum transfers if their fiscal capacity exceeds their fiscal need, which was the case for just four Saxon municipalities in 2002. All the others received lump-sum transfers, partly equalising the gap between their fiscal need and fiscal capacity.

In addition to lump-sum transfers, the Saxon fiscal transfer system provides for further unconditional transfers to take account of fiscal needs independently of the municipality's fiscal capacity; it dedicates specified amounts for transfers to compensate for burdens linked to roads, cultural amenities, and snowfall, and includes earmarked transfers. Before presenting the models below, a suitable indicator is developed for including protected areas in the Saxon fiscal transfer system.

4.3.2 Conservation units as an indicator

Conservation units (CUs) are used as an indicator for taking account of designated protected areas in the Saxon fiscal transfer scheme. Conservation units are standardised areas within the boundaries of a municipality that belong to one of the following categories of protected areas according to Saxon nature conservation law: national park, special area of conservation (SAC) according to the *EU Habitats Directive*[2], special protection area (SPA) according to the *EU Birds Directive*[3], nature reserve, biosphere reserve, nature park, and landscape reserve. These categories are defined by German nature conservation law. Data on protected area boundaries are available from conservation authorities. Hence, CUs represent a simple yet comparable indicator capable of being included in the communal fiscal transfer systems of the German states in various ways, thus reflecting German federalism. For the standardisation of protected areas, a conservation weight is introduced, taking account of the conservation value of the management category in question (Table 4.2). From an economic perspective, the conservation weight also takes into account – in other words rewards – spillover benefits associated with categories of international and European significance, such as national parks, biosphere reserves, and the European Natura 2000 network. Land-use restrictions associated with the various management categories are also considered.[4] For example, national parks are of very high conservation value; they represent a management category of international significance and are associated with the strictest land-use restrictions in German nature conservation law. For this reason, one hectare of national park corresponds to one CU, setting the reference value for the other categories. In contrast, landscape reserves are usually of regional impor-

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2 EU Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora.

3 EU Directive 79/409/EEC on the conservation of wild birds.

4 The conservation weights applied here were estimated with the assistance of Klaus Henle, Head of Department of Conservation Biology at the Helmholtz Centre for Environmental Research – UFZ. In practice, conservation authorities may decide on weights according to the conservation value of different protected areas. A scientific justification may also include an analysis of land-use restrictions and associated opportunity costs as well as an economic valuation of spillover benefits.

tance with relatively low land-use restrictions. Thus, one hectare of landscape reserve is assumed to correspond to 0.3 CU.[5]

Table 4.2: Estimated conservation weights for different categories of protected areas.

Management category	Conservation weight
National park	1
Special area of conservation (SAC), EU Habitats Directive	0.9
Special protection area (SPA), EU Birds Directive	0.9
Nature reserve	0.7
Biosphere reserve	0.6
Nature park	0.4
Landscape reserve	0.3

The CUs of each municipality are identified by overlaying GIS[6] layers of the various categories of protected areas over a total of 537 municipal borders in Saxony, while avoiding double counts.[7] Figure 4.2 shows all protected areas in Saxony as of early 2004 superimposed on its municipal borders as of January 1, 2002.[8] 41 percent or approximately 758,000 hectares of Saxony's total area are designated under nature conservation law (Table 4.3). Due to the conservation weights applied, this corresponds to approximately 365,000 hectares CU. Compared to Chemnitz and Leipzig, the administrative district of Dresden has a higher percentage of CUs due to having large protected areas, including Saxony's single national park and biosphere reserve.

To simplify matters, the models presented include CUs only as a quantitative indicator. As a second step, both the quantity and the quality of protected areas should be considered in order to prevent municipalities striving to accumulate low quality protected areas.

5 If all protected areas were to be considered in the same way, all the conservation weights would be equal to one. The sum of a municipality's CUs would then be the sum of all designated protected areas within its territory.

6 Geographical information system.

7 An area designated as a nature reserve, which also lies in a biosphere reserve, is to be counted once using the area and conservation weight of the highest category associated with it.

8 GIS data for all types of protected area in Saxony became publicly available only in 2004.

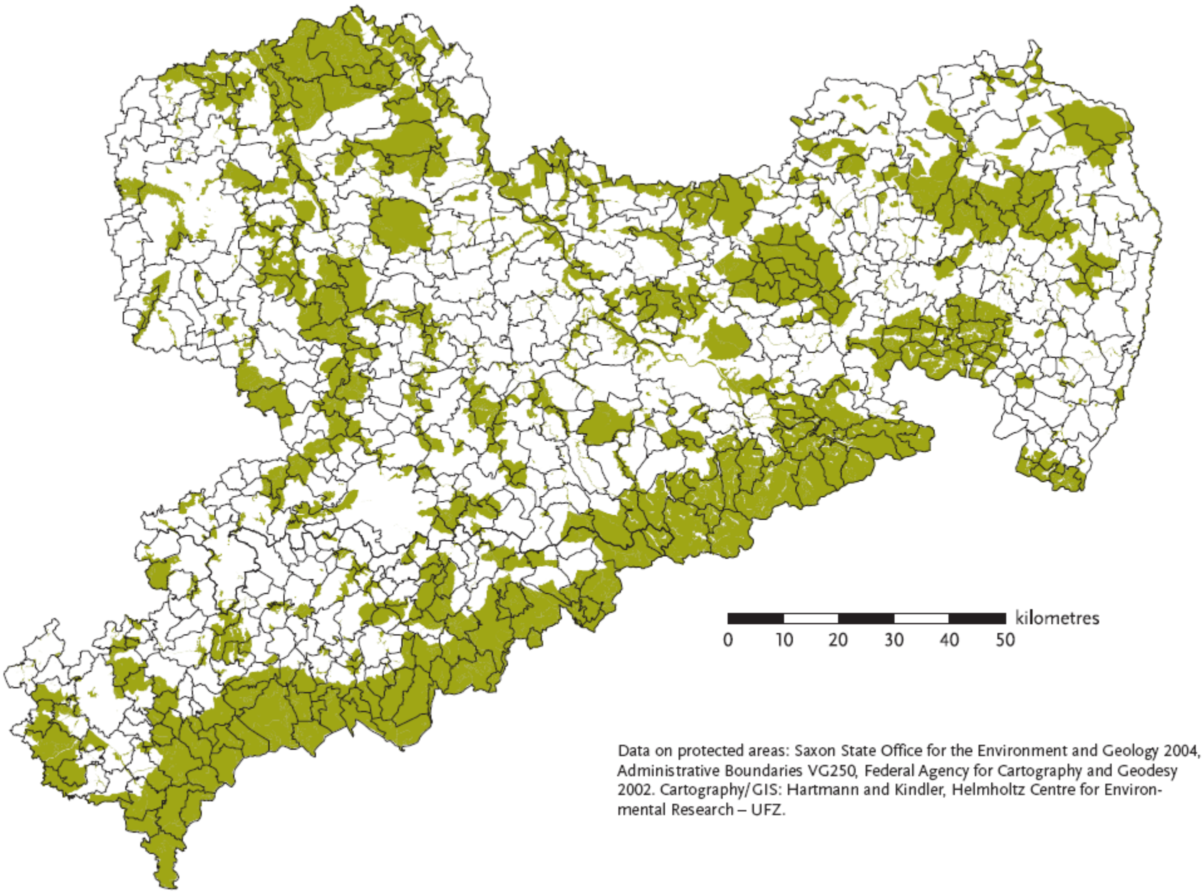

Figure 4.2: Protected areas overlaid over municipal borders in Saxony.

Table 4.3: Inhabitants, protected areas and conservation units (CUs) in Saxony.

Administrative district	Inhabitants	Total area (ha)	Protected area (ha)	Protected area (%)	CUs (ha)	CUs (%)	CUs per inhabitant
Chemnitz	1,621,284	613,007	270,770	44.2	118,289	19.3	0.07
Dresden	1,712,562	796,944	322,094	40.4	168,080	21.1	0.10
Leipzig	1,091,735	440,669	165,439	37.5	78,989	17.9	0.07
Saxony	4,425,581	1,850,620	758,303	41.0	365,359	19.7	0.08

4.3.3 Model 1: Including conservation units in general lump-sum transfers

In the first model, protected areas are taken into account by creating a further additional approach – the "conservation approach" – in the distribution of lump-sum transfers, thereby increasing fiscal need while acknowledging local fiscal capacity. For this purpose, CUs need to be converted into the generic indicator of inhabitants. This procedure mirrors the existing consideration of area in fiscal transfer systems in other German *Länder*[9]. In the model calcu-

[9] For example Brandenburg: additional approach with ten inhabitants per square kilometre area for districts; Saxony-Anhalt: principal approach with 15 inhabitants per square kilometre area for districts; Rhineland-

lation presented, we assume that a political decision is taken to make one hectare CU equal to one inhabitant (average population density in Saxony in 2002: 2.36 inhabitants per hectare). Thus, the “overall approach” of a municipality is given by the sum of weighted inhabitants and schoolchildren plus its CUs. In this way, the fiscal need of a municipality is increased according to the CUs within its municipal boundaries.

Ideally, the equivalence between CUs and inhabitants should be justified empirically by estimating the opportunity costs of protected areas and the municipality’s direct conservation management costs, in order to approximate the “actual” fiscal need. It is very difficult (if not impossible), however, to develop an indicator and an appropriate weighting procedure that “correctly” reflects the fiscal need of a municipality. A similar problem is encountered when using weighted inhabitants to approximate municipal fiscal needs, a (controversial) argument developed in the literature on public finances in the 1930s (Brecht 1932). Rhineland-Palatinate and Schleswig-Holstein have already abolished the weighting of inhabitants, in contrast to the prevailing practices of other German states.

In the present model, the base amounts calculated for district-independent cities (1,144.21 euros) and for communities (478.04 euros) are lower than those obtained in the original fiscal transfer system in 2002 (see above). Hence, municipalities with no or few CUs receive fewer lump-sum transfers, while others make gains according to their CUs. Due to the differing base amounts, one weighted inhabitant or schoolchild, as well as one hectare CU, more than doubles the fiscal need of a district-independent city compared to the smaller communities. From an economic point of view this is justified by the higher opportunity costs of protected areas in large cities.

Figure 4.3 illustrates the percentage changes in general lump-sum transfers due to CUs for each Saxon municipality, compared to lump-sum transfers originally received in 2002. On the winners’ side, most municipalities increase their lump-sum transfers by up to 25 percent, but there are some municipalities that more than double them. The ten municipalities ranking highest are located within one of the large protected areas in Saxony’s peripheral regions. Half of them lie in the biosphere reserve Oberlausitzer Heide- und Teichlandschaft, while the others are part of the nature parks Dübener Heide and Erzgebirge-Vogtland, as well as the national park Sächsische Schweiz. The absolute increase in lump-sum transfers of these ten municipalities ranges between 700,000 and 1.5 million euros. By contrast, the vast majority of municipalities in Saxony lose only up to 25 percent of their lump-sum transfers (Table 4.4).

Palatinate: additional approach with two inhabitants per square kilometre area above state average. In protected area statistics at local and state levels, the unit hectare (one hectare equals 0.01 square kilometre) is more common and henceforth used to incorporate CUs into fiscal transfers.

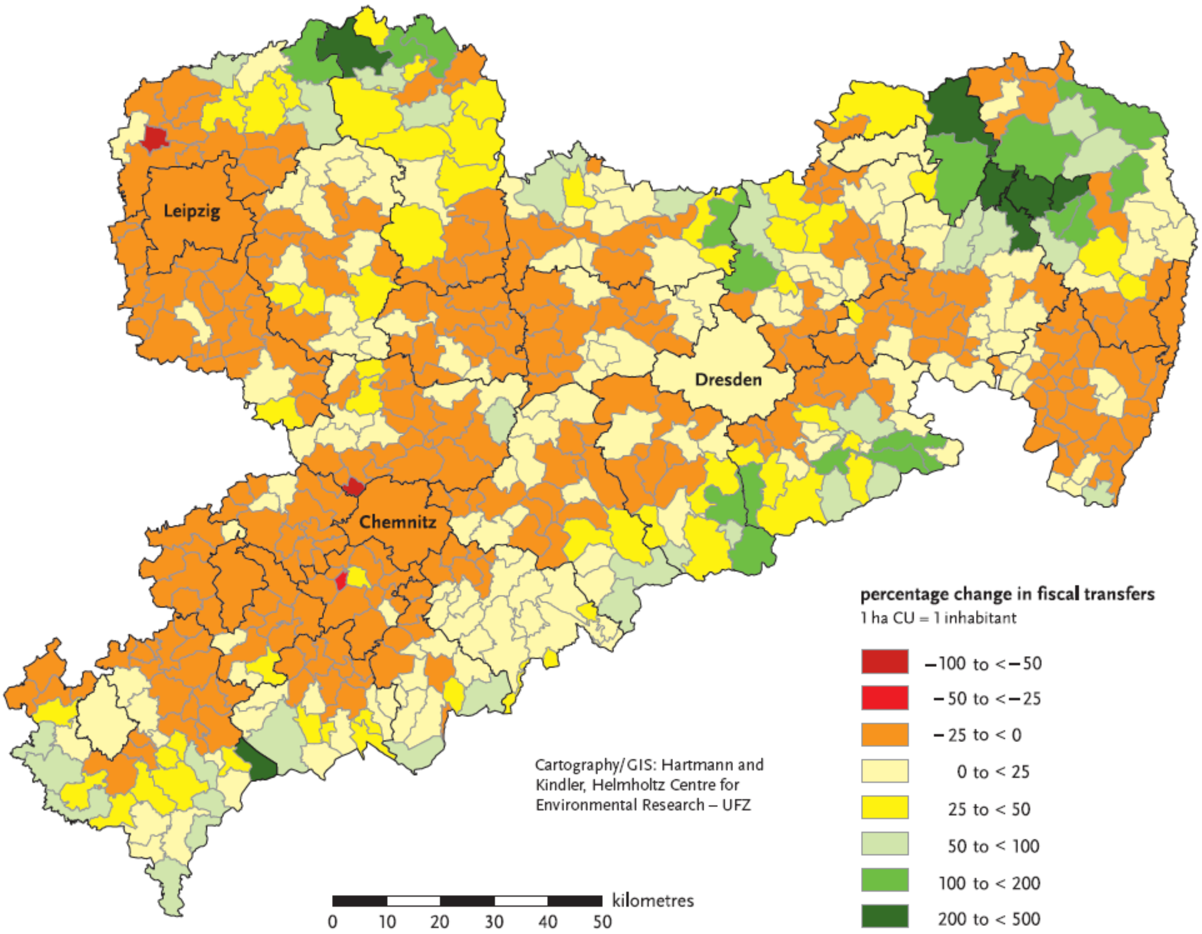

Figure 4.3: Model 1: Percentage change in general lump-sum transfers when the Saxon fiscal transfer system 2002 was expanded to include designated protected areas. In this model, conservation units (CUs) are used in addition to inhabitants and schoolchildren to calculate the fiscal need of a municipality, assuming one hectare CU is equal to one inhabitant.

Table 4.4: Model 1: Distribution of winning and losing municipalities across Saxony if the Saxon fiscal transfer system 2002 included designated protected areas.

	Percentage change in general lump-sum transfers							
Administrative district	< –50	–50 up to < –25	–25 up to < 0	0 up to < 25	25 up to < 50	50 up to < 100	≥ 100	Number of municipalities
Chemnitz	1	1	113	64	21	13	1	214
Dresden	0	0	99	65	21	16	19	220
Leipzig	1	0	61	18	15	4	4	103
Saxony	2	1	273	147	57	33	24	537

4.3.4 Model 2: Devoting a specified amount to ecological fiscal transfers

The second way of including CUs in communal fiscal transfers is designed in analogy to the *ICMS Ecológico* in Brazil (see Box 4.1). This fiscal instrument has been implemented by several Brazilian states since the early 1990s (Grieg-Gran 2000; May et al. 2002; Ring 2008b). Ecological fiscal transfers are determined by multiplying the ecological index of a municipality by a specified amount of money devoted to conservation. For a given quantity of overall CUs in the state, each municipality's ecological index is controlled by its Municipal Conservation Factor (MCF), representing the share of municipal CUs in relation to its total area. In this way, the relative land-use restrictions associated with protected areas are compensated, and a higher CU to municipal area ratio will lead to an increased absolute amount of ecological fiscal transfers. Thus, a municipality with a MCF of 90 percent receives more ecological fiscal transfers than one with a MCF of only 30 or 50 percent.

Applied to the situation in Saxony, a specified amount is devoted to unconditional fiscal transfers based on CUs, in analogy to Saxon fiscal transfers for the compensation of burdens associated with road maintenance (about 90 million euros each year), provision of cultural services (about 30 million euros), and the removal of excess snow (SMF 2007). For the purpose of illustration, we assume that 90 million euros are devoted to conservation services, similar to the amount used annually for roads. This is about 2.7 percent of the 3,283 billion euros available for communal fiscal transfers in 2002. By comparison, Brazilian states devote 0.5 to seven percent to CU-based fiscal transfers (Ring 2008b).

Figure 4.4 and Table 4.5 present the percentage changes in fiscal transfers to Saxon municipalities for the second model. Results are given in net changes in fiscal transfers, consisting of additional ecological fiscal transfers due to CUs and reduced lump-sum transfers.

> **Box 4.1: International experiences with fiscal transfers for protected areas**
>
> Both Brazil and, more recently, Portugal have implemented ecological fiscal transfers, compensating municipalities for land-use restrictions imposed by protected areas. The ICMS Ecológico has been adopted by 12 out of 27 Brazilian states; others are preparing relevant legislation (Ring 2008b).
>
> Conservation units (CUs) are the ecological indicator used by all states. They are defined according to the National System of Conservation Units, and relate to the categories of protected areas for biodiversity conservation in Brazil. Paraná was the first state in 1992 to introduce ecological indicators for the redistribution of state value-added tax income to municipalities: 2.5 percent of the amount to be distributed to the local level is allocated according to CUs; another 2.5 percent considers water protection areas within a municipality's territory (May et al. 2002).
>
> As of January 1, 2007, Portugal has a new national community financing law that includes ecological fiscal transfers. The new fiscal transfer scheme explicitly rewards municipalities for designated Natura 2000 sites and other protected areas within their territories (de Melo and Prates 2007).

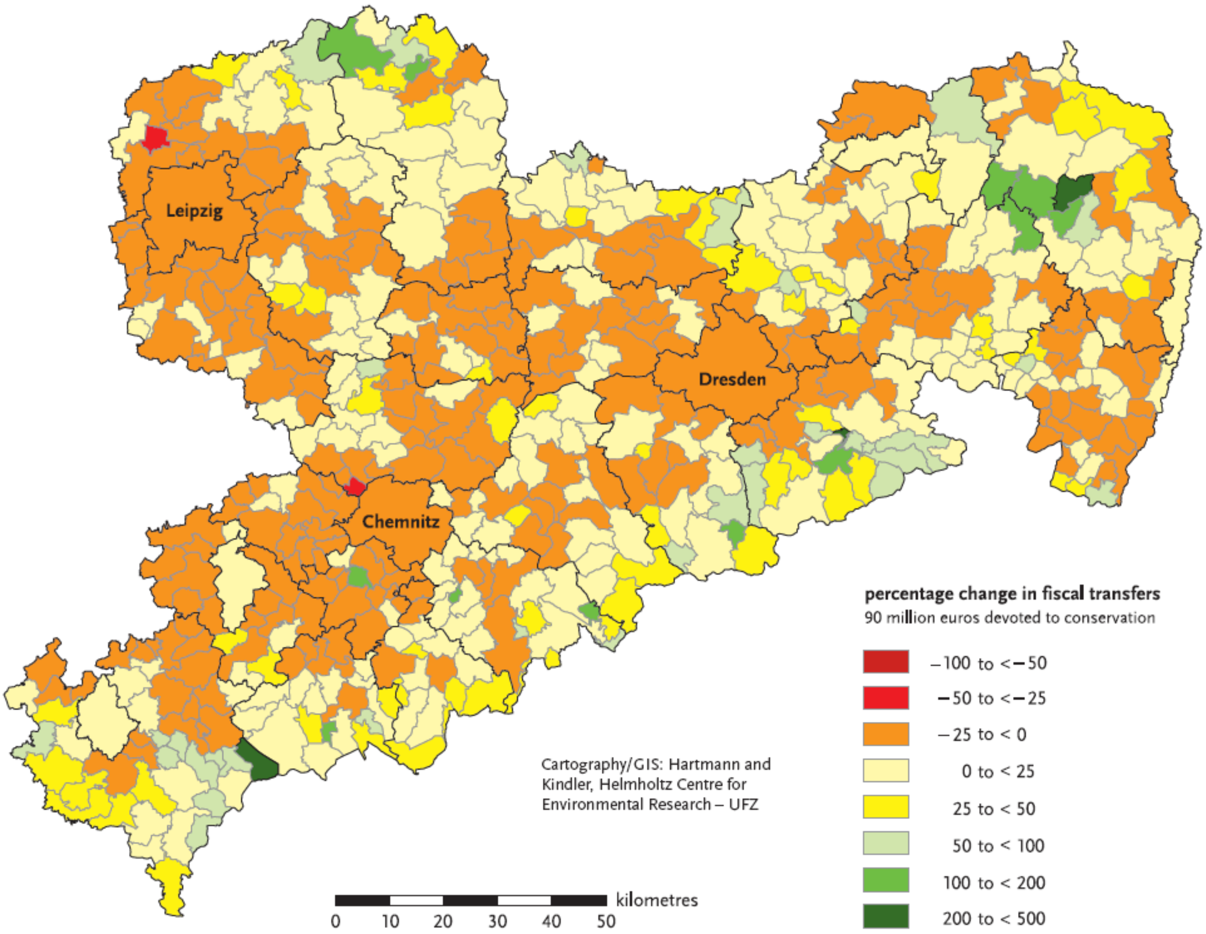

Figure 4.4: Model 2: Percentage change in fiscal transfers when 90 million euros were devoted to conservation in the Saxon fiscal transfer system in 2002. The map indicates net changes in fiscal transfers consisting of additional ecological fiscal transfers, based on conservation units (CUs) in relation to the total municipal area, and reduced lump-sum transfers.

Table 4.5: Model 2: Distribution of winning and losing municipalities across Saxony if the Saxon fiscal transfer system devoted 90 million euros to conservation.

	Percentage change in net fiscal transfers							
Administrative district	< –50	–50 up to < –25	–25 up to < 0	0 up to < 25	25 up to < 50	50 up to < 100	≥ 100	Number of municipalities
Chemnitz	0	1	93	75	27	13	5	214
Dresden	0	0	84	82	28	18	8	220
Leipzig	0	1	58	32	8	2	2	103
Saxony	0	2	235	189	63	33	15	537

The monies devoted in advance to CUs are no longer available for lump-sum transfers, reflecting a budget-neutral fiscal reform. Similar to the first model, most municipalities either gain or lose up to 25 percent of their fiscal transfers. However, in the present simulation, with 90 million euros devoted to conservation, 39 municipalities move from the losing to the winning side, while fewer municipalities double their income from fiscal transfers.

4.4 Which option to choose?

In principle, both models are suitable for including protected areas in intergovernmental fiscal transfers to the local level. The indicator "conservation units" offers further possibilities for greening communal fiscal transfers. A specified amount per hectare CU could be allocated to municipalities, in the way Portugal amended its communal financing law in 2007 (see Box 4.1). In the end, the choice among these options is a political decision, as is the choice of conservation weights and funds reserved for conservation purposes. Although there are theoretical and scientific arguments to support the different options, political reasoning as well as community lobbying strongly influence the specific design of a state fiscal transfer scheme. This is exemplified in current practices of including area or recreational functions in German communal fiscal transfer laws (Ring 2002).

There is a basic difference, however, between the two models presented. In the first model, municipalities only benefit if they receive lump-sum transfers. If their fiscal capacity still exceeds fiscal need despite including CUs in the fiscal transfer system, it is assumed that the financial status is healthy enough to cope with conservation-related direct and opportunity costs. By contrast, the second model always provides for municipal income for CUs irrespective of fiscal capacity. So the two models differ in the question of whether protected areas and associated fiscal needs should be valued in relation to or irrespective of fiscal capacity. Many national parks regions benefit economically from higher tourism income. Taking account of CUs in the distribution of general lump-sum transfers would better account for municipal income (local taxes) generated through attractive tourism destinations. At the same time, less attractive areas of high conservation value would be able to increase their income based on CUs, allowing the latter to provide basic municipal services for the remaining inhabitants.

At any rate, existing experiences with ecological fiscal transfers in Brazil have shown that their implementation should be accompanied by a sound information policy. Otherwise, municipalities may simply not know that protected areas generate municipal income for them (Grieg-Gran 2000). Nevertheless, local preferences differ, and there may be municipalities that – despite ecological fiscal transfers – are not interested in biodiversity conservation (May et al. 2002), while others are intrinsically motivated in conservation policies irrespective of economic incentives.

Two important developments should be mentioned with regard to Saxony: *First,* a new administrative reform is under way that involves retaining only the three district-independent cities of Leipzig, Dresden, and Chemnitz (Sächsische Staatsregierung 2007). When the plans are realised in mid-2008, general lump-sum transfers for the cities of Görlitz, Hoyerswerda, Plauen, and Zwickau will be much lower. The change of status will be hard for these cities, irrespective of the consideration of protected areas. A transition phase has already been planned to assist them in adapting to fewer lump-sum transfers. *Second,* the models presented include Saxony's protected areas as of early 2004. Since then, a significant number of SPA sites according to the EU Birds Directive have been reported to the EU Commission, constituting about 13.5 percent of the Saxon state area, compared to approximately four percent incorporated in the models. Including these new reserves would mean increased transfers for a number of municipalities due to the high conservation weight associated with this category.

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5 Integrating local ecological services into intergovernmental fiscal transfers: The case of the ecological ICMS in Brazil

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Abstract

Local conservation efforts are often related to benefits at higher governmental levels. On the one hand, these efforts are strongly connected to local land-use decisions. On the other hand, activities such as sustainable water management or biodiversity conservation are associated with regional, national or even global public goods. Therefore, spatial externalities or spillovers exist, which – if not adequately compensated for – lead to an underprovision of the public goods and services concerned. This article investigates intergovernmental fiscal transfers as an innovative instrument for compensating local jurisdictions for the ecological goods and services they provide across local boundaries. From a public finance perspective, fiscal transfers are a suitable instrument for internalising spatial externalities. However, most federal states use this instrument predominantly for social and economic public sector functions rather than for ecological ones. This article investigates the case of the ecological “ICMS” that was first introduced by a few states in Brazil during the 1990s. Part of the revenue from this value-added tax is redistributed to the local level on the basis of ecological indicators. In this way, the state level uses fiscal transfers to compensate municipalities for the existence of protected areas and other ecological services provided within their territories. The Brazilian experience illustrates that such fiscal transfers can represent both a compensation for land-use restrictions and an incentive to value and engage in more conservation activities at the local level.

Keywords:

Fiscal transfers, environmental federalism, conservation benefits, spillover effects, local ecological services, Brazil

5.1 Introduction

Sustainable land use requires a variety of conservation efforts and services, not least at the local level. There is a close link between the environmental quality of a landscape, its land-use pattern and the type of management performed by public jurisdictions and private land users. However, local communities can often obtain greater benefits from activities that contribute to the loss of biodiversity and ecosystem services. Providing incentives for conservation efforts in the form of benefits for local people has proven to be very difficult (Millennium Ecosystem Assessment 2005a, p. 92ff). There are few incentives for local actors to engage in conservation activities, especially when ecological benefits cross local boundaries (Perrings and Gadgil 2003). In Europe, compensation payments for ecological services provided by private land users are being introduced more and more widely, mainly in the context of co-funded EU agri-environmental programmes. What is largely lacking in European member states, however, are economic instruments that explicitly address local public actors in their role as providers of ecosystem services over the long term. Local governments usually seek to achieve socio-economic development by attracting both industries and residents as tax payers. Municipalities are rarely able to generate income from the provision of ecological services; on the contrary, they are often restricted in their sovereignty, e.g. regarding land-use planning. This is the case with a number of services, such as water protection or nature reserves. Decisions on the designation of the protected areas concerned are often taken by institutions above the local level, whereas the concrete consequences in terms of restrictions in land use are borne by local actors, often without any (or adequate) compensation.

This article describes an innovative instrument that is able to address this basic problem. Intergovernmental fiscal transfers are analysed with regard to their potential within federal systems for compensating for ecological services provided by local governments. From a public finance perspective, it is the ‘value added’ of local ecological services, i.e. the benefits that cross the boundaries of local jurisdictions, which are of special interest. The first part of this article gives a brief introduction to public finance principles as they relate to ecological goods and services.[1] The main part presents a case study from Brazil, focusing on the status quo and the incentive effects of integrating ecological public sector functions into intergovernmental fiscal transfers. Since the 1990s, several states in Brazil have introduced fiscal transfers that explicitly compensate municipalities for certain ecological services, such as watershed protection and conservation areas (Bernardes 1999; Grieg-Gran 2000; Loureiro 2002; May et al. 2002). The various ways of introducing ecological indicators into the Brazilian system of intergovernmental fiscal transfers, as well as the experiences gained so far, are analysed and important criteria for success elaborated. Finally, the prospects for transferring this instrument to other federal systems are explored.

[1] For a more detailed interpretation of public finance principles with respect to ecological public sector functions and their relevance to European countries, see Ring (2002).

5.2 Compensating for the value added of local ecological services

5.2.1 Principles of fiscal federalism

Fiscal federalism is a sub-field of public finances that explores the roles of the different levels of government and their interrelationships through fiscal instruments. Its major task is one of effectively and efficiently allocating public sector functions, expenditures and revenues to the central, state and local governmental levels in federal systems. As Oates (1999, p. 1120) puts it, "... we need to understand which functions and instruments are best centralised and which are best placed in the sphere of decentralised levels of government". Regarding the allocation function of the public sector, the principle of fiscal decentralisation has been advanced (Musgrave 1959; Oates 1972). Most public goods and services are provided more efficiently when production and consumption are limited to the lowest governmental level possible. This makes it possible to respond more appropriately to the regionally differing preferences of the population (Tiebout 1956).

The decentralisation rule for allocating public goods and services applies in the absence of economies of scale. Given economies of scale, the provision of the public goods and services concerned should be moved to the cost-efficient centralised level (Postlep and Döring 1996). Due to the characteristics of non-rivalry and non-excludability of many public goods, spatial externalities or spillovers may exist between jurisdictions. In this case, the principle of fiscal equivalence comes into play, which advocates achieving a match between those who decide about a collective good, those who pay for it and those who receive its benefits (Buchanan 1950; Olson 1969). Social welfare is increased through the differentiation of public services in accordance with costs and preferences at the appropriate governmental level. The realisation of fiscal equivalence may require competencies to be shifted to a more centralised level of government. Regional cooperation, e.g. by way of negotiations between the relevant parties, can also lead to an efficient Coasian type of resolution of jurisdictional spillovers (Bergmann 1999; Oates 2001). A third option has been mentioned by Olson (1969). Provided diseconomies of large-scale operations require local provision, spillovers can be internalised through fiscal transfers from more centralised levels to the local level. In this way, the local government is compensated for the external benefits of its expenditures. We focus on this potential of fiscal transfers to internalise spillover benefits to other jurisdictions. In the next section, we discuss how the principles of fiscal federalism apply to ecological goods and services.

5.2.2 Environmental federalism and spatial externalities

What are the consequences of the decentralisation rule and the principle of fiscal equivalence for environmental issues? The decentralisation rule implies that the provision of ecological goods and services should be assigned to lower levels of government where appropriate. However, the execution of this rule calls for a differentiated approach. Appropriate solutions have to be sought according to the specific characteristics of the various natural resources and environmental compartments. This is reflected in ongoing discussions regarding the competencies of the national or even supranational governmental level versus the state or local level in environmental standard setting (Döring 1997; Oates 1998, 1999).

Public goods that tend to be underprovided at decentralised levels are basic and applied research, including that concerning the development of environmental policy instruments, but also the dissemination of information on harmful environmental impacts or the development of pollution control techniques. These issues need to be assigned to a more centralised level of government (Oates 2001). Global change problems such as climate change or biodiversity loss also point to the need for a fundamental role of central levels of government (Ring 2004). Highly mobile environmental compartments and associated pollutants that easily cross administrative boundaries create far-reaching spatial externalities. The emissions of carbon dioxide and other air pollutants associated with climate change and the depletion of the ozone layer require more centralised, if not global, emission policies (Hansjürgens 1996).

In contrast, environmental policy associated with less mobile environmental compartments and spatially restricted problems is better suited for assignment to decentralised levels of government (Ring 2002). This is due to the lower probability of causing spatial externalities. Tasks relating to land use and soil conservation, as well as public functions associated with inland waters, can usually be solved within national boundaries. Despite the general suitability of many land-use issues for being assigned to lower governmental levels, spatial externalities may require different, more appropriate solutions.

Regarding water resources, such solutions may be necessary in relation to public policies for pollution control related to transboundary spillovers. In the context of water pollution in the US, for example, Sigman (2005) estimates the environmental costs generated downstream due to free riding states when rivers cross state boundaries. Whereas transboundary water pollution is associated with negative spillovers, priority areas for water protection can involve positive spillovers. In contrast to certain local costs, be it in terms of land-use restrictions or measures for keeping up and improving the quality of the respective reserves, the benefits from some of these activities cross local boundaries. For example, water protection zones are often located in rural areas, mostly providing drinking water far beyond local demand. Urban agglomerations and capital cities with high population densities and industrial activities rely especially heavily on water resources lying outside their own municipal boundaries. In the case of water resources, an important task consists in properly valuing these resources and their functions which should then, as far as possible, be reflected in water prices. In the European Union, the Water Framework Directive now obliges member states both to develop management regimes across state and national boundaries according to the spatial dimensions of river basins and to find solutions to fully reflect resource costs in water prices (Hansjürgens and Messner 2002; Unnerstall 2006). However, for various reasons, resource costs are not yet fully reflected in prices; indeed, with regard to certain tasks related to long-term resource protection, this may not even be a feasible solution.

The conservation and sustainable use of biodiversity is another example of the widespread existence of spatial externalities (Perrings and Gadgil 2003; Ring 2004). The loss of biodiversity and the threat to services provided by ecosystems for human well-being are among the serious global change problems, demanding in many cases centralised standard setting and policies. This is reflected in numerous international conventions regarding biological diversity and species protection. In the European Union, the Habitats and Birds Directives set strict standards to be implemented by member states at national and state levels (Similä et al. 2006). Despite centralised standards, decentralised activities related to local land use – seen cumulatively – have a tremendous influence on the state of biodiversity worldwide. Compared to water resources, ensuring that the value of ecological services is reflected in market prices is even more difficult, if not impossible, for many areas of biodiversity conservation (Gowdy 1997; Millennium Ecosystem Assessment 2005b). This is especially true for benefits related to non-use values, such as existence and option values that may accrue to people everywhere. For example, Horton et al. (2003) attempted to estimate the non-use values for a programme of protected areas in the Brazilian Amazon by eliciting individual preferences in Italy and the UK. Although the contingent valuation study exemplified the difficulties and uncertainties of such a global approach, it clearly indicated that the majority of households in Italy and the UK were willing to pay to support large-scale tropical forest preservation efforts. Hence, large-scale positive spillovers exist at least for certain public goods of global relevance. The practical consequences of spatial externalities related to species protection are illustrated by an empirical study by List et al. (2002). In their study of federal and state spending under the Endangered Species Act in the US, they identified the phenomenon of free riding on the part of the states. States tend to spend less (relative to the federal government) on those species that demand a large habitat area and whose preservation causes conflicts with economic development. Perrings and Gadgil (2003) address a number of reforms necessary to reconcile both local and global public benefits of biodiversity conservation. One of them is adjusting incentives to allow local communities to be rewarded and paid for their conservation efforts (Ring 2002; Unnerstall 2004; Millennium Ecosystem Assessment 2005b).

In the following case study, the focus for solving such discrepancies will be on fiscal transfers. Spillovers associated with the provision of public goods and services at the local level can be internalised through intergovernmental fiscal transfers from more centralised levels to the local level. These transfers may compensate municipalities for the external benefits of their conservation expenditure, as well as for their opportunity costs related to land-use restrictions to be borne. This is especially necessary in relation to social benefits that accrue in the long term, where public actors are bearing the costs arising in the present. In this way, the 'value added' of local ecological goods and services is acknowledged socially, which at the same time can provide an incentive for local actors to engage in more conservation activities.

5.3 Fiscal transfers for local ecological services in Brazil

Brazil has 27 states, each with an elected government which has revenue-raising powers. The ICMS tax (*Imposto sobre Circulação de Mercadorias e Serviços*) represents the largest source of state revenue in Brazil, constituting approximately 90 % of overall state tax revenues (Loureiro 2002). It is also an important source of revenue for local governments. The ICMS is a tax on goods and services, similar to the value-added taxes in other countries. It is collected on commercial transactions and exchanges of goods and services, such as energy, transportation and communication (May et al. 2002). The Federal Constitution of Brazil as adopted in 1988 decrees that 25 % of the revenues raised by this tax are to be allocated by the state to the local level of government. Constitutional law further stipulates that 75 % of the total amount passed on to the municipalities is to be distributed in accordance with the share of the state ICMS that has been collected within that municipality. The state governments determine the indicators to be used for allocating the remaining 25 %. Typical indicators are based on population, geographical area and primary production (Grieg-Gran 2000). Since the 1990s, the states began to introduce ecological indicators.

Paraná was the first state to introduce the ICMS Ecológico (ICMS-E) (May et al. 2002), meaning the ICMS along with an allocation of tax revenues based on environmental indicators.[2] In 1990 and 1991, the relevant laws and implementing regulations were adopted that allowed for consideration of ecological indicators in the ICMS (Loureiro 2002). Drawing on the experience of Paraná, which started using ecological indicators in 1992, the states of Minas Gerais (1996), São Paulo (1996), Rondônia (1997), Mato Grosso do Sul (2002), Tocantins and Pernambuco (2003) started operating a similar system a few years later (Grieg-Gran 2000; Loureiro 2002; May et al. 2002; CPRH 2003). The state of Rio Grande do Sul passed an ICMS-E Law in 1993, followed by implementing laws and regulations in 1997 and final adoption in 1999 (Freitas 1999). ICMS-E legislation also exists in the states of Amapá (1996), Mato Grosso (2001) and, very recently, Rio de Janeiro (2007). Other Brazilian states, namely Santa Catarina, Espírito Santo and Goiás, drafted ICMS-E legislation; Amazonas, Bahia and Ceará submitted ICMS-E legislation to their respective state legislatures (Bernardes 1999; Freitas 1999; Leite 2001; Loureiro 2002; MMA 2002; Arantes 2006). Discussions are being conducted in the latter states, but so far ecological ICMS legislation has been met with opposition, mostly due to fiscal competition between municipalities for scarce revenues.

5.3.1 Paraná: valuing watershed protection areas and conservation units

Following the implementation of more strict environmental legislation in the early 1980s, a number of municipalities in Paraná that had protected areas on their territory exerted pressure on the state legislature and government agencies (Loureiro 1998). The land-use restrictions associated with large conservation and watershed protection areas were preventing the municipalities from developing productive activities and thereby generating value added. The municipality of Piraquara is a typical example of this situation: 90 % of municipal territory consists of designated protected areas for conserving a major watershed to supply the Curitiba

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[2] The ICMS Ecológico is also known as the "ecological value-added tax". This holds both for scientific literature and common language use in Brazil. From a public finance perspective, however, this term is misleading. An ecological tax would be a tax whose assessment base is related to ecological indicators. The ICMS Ecológico, by contrast, uses ecological indicators for the allocation of its revenues. Therefore, economically speaking, the term ecological fiscal transfer is more appropriate and will be used henceforth in this article.

metropolitan region (1.5 million inhabitants) with drinking water, and the remaining 10 % occupies protected areas for biodiversity conservation (May et al. 2002).

The ICMS Ecológico was introduced in response to these concerns, as an instrument to compensate municipalities with large protected areas for the land-use restrictions they faced, while providing incentives for conservation (Loureiro 2002). In the case of Paraná, state decision-makers considered the long-term costs of water treatment needed due to uncontrolled development around water sources. They also were worried about the serious deterioration of the state's land cover with respect to biodiversity protection (Loureiro, cited in Echavarría 2000). Following the adoption of the Ecological ICMS Law in 1991, 5 % of the total amount distributed to the local level has been based on ecological indicators since 1992. Half of this (2.5 %) is distributed to municipalities that have watershed protection areas on their territories which partly or completely provide services for public drinking water systems in neighbouring municipalities. The other half is for those municipalities that have "conservation units" (Loureiro 2002). Conservation units (*CUs*) are conservation areas that consist of completely protected and restricted sustainable use areas that can be publicly managed (federal, state or municipal level), privately owned or managed by public-private partnerships. The ICMS-E revenue accrues to the municipality and not to the owner of the land. Therefore, the incentive effect primarily addresses local public authorities. However, as we will see later, there is also an incentive effect to encourage public-private partnerships in terms of more environmentally sound land uses. The protected areas may be used indirectly (biological reserves, ecological stations and parks) or directly (indigenous areas, extractive reserves and sustainably managed forests). In either case, they have to be registered and legally defined in order to be considered for ICMS-E allocation (Grieg-Gran 2000). In Paraná, the ICMS-E programme is administered by the Paraná State Environmental Institute (*Instituto Ambiental do Paraná, IAP*).

5.3.2 Synopsis of existing approaches

The states of Paraná, Minas Gerais, Rondônia and São Paulo have now been operating the ICMS-E for several years. Mato Grosso do Sul, Pernambuco and Tocantins introduced the new system only recently. Each state is free to decide upon the indicators for distributing 25 % of the ICMS to the local level, and therefore, different operating systems are in place. This is important to bear in mind when comparing various effects among the states. The indicator "conservation units" has been introduced by all states with ICMS-E legislation. Although the states use slightly different methods to calculate the ecological index of a municipality, the basic procedure is the following:

The revenues allocated are based on the ecological index EI_i of municipality i multiplied by the total amount of ICMS-E revenues dedicated to conservation units. The ecological index EI_i is calculated by dividing the municipal conservation factor MCF_i by the state conservation factor SCF. For each municipality i $(i = 1,\ldots, z)$ the ecological index EI_i can be written as

$$EI_i = \frac{MCF_i}{SCF} . \qquad (1)$$

EI_i is the ecological index of municipality i, MCF_i the municipal conservation factor of municipality i, SCF the state conservation factor, and z the total number of municipalities in the state.

The municipal conservation factor MCF_i is based on the total area set aside for protection in terms of conservation units CU in relation to the total area of the municipality:

$$MCF_i = \frac{Area\ CU_i}{Area\ M_i} . \tag{2}$$

$Area\ CU_i$ is the total area of conservation units in municipality i and $Area\ M_i$ the total area of municipality i.

The CUs of a municipality are calculated according to Eq. (3) where the protected areas are weighted according to the different categories of management. Table 5.1 shows the conservation weights CW_n for different types of protected areas in Paraná.

If n denotes the different categories of management,

$$Area\ CU_i = \sum_n protected\ area_n \times CW_n . \tag{3}$$

CW_n is the conservation weight for management category n.

The state conservation factor SCF is given by the sum of all municipal conservation factors in the state:

$$SCF = \sum_{i=1}^{z} MCF_i . \tag{4}$$

Table 5.1: Conservation weights CW_n for different management categories n of protected areas in Paraná.

Management category	Conservation weight
Ecological research station	1.0
Biological reserve	1.0
Park	0.9
Private natural heritage reserve	0.8
National, state or municipal forest	0.7
Indigenous area	0.5
Environmental protection area I	0.1
Area of relevant ecological interest	0.1
Special, local areas of tourist interest	0.1
Buffer zones	0.1

Source: Adapted from Grieg-Gran (2000).

Paraná was also the first state to evaluate the quality of protected areas and to include this in the calculation of the ecological index (Grieg-Gran 2000; Loureiro 2002; May et al. 2002). The additional quality index of each protected area is assessed by regional officers of the state environmental agency on the basis of variables such as physical quality, biological quality (fauna and flora), quality of water resources (within the *CU* and in its surroundings), physical representativeness, and quality of planning, implementation and maintenance.[3] Minas Gerais (2005) planned for a quality factor early on, but only introduced the respective regulation in 2005. Although more states with ICMS-E legislation call for a quality index, in most cases they are only partially implemented, if at all.

Table 5.2: ICMS allocation for ecological indicators in the states operating the ICMS Ecológico.

State	ICMS-E legislation adopted	Ecological share of total ICMS (%)	Allocation to indicators (%)	Ecological indicators
Paraná	1991	5	2.5	Watershed protection areas
			2.5	Conservation units
São Paulo	1993	0.5	0.5	Conservation units
Minas Gerais	1995	1	0.5	Conservation units
			0.5	Solid waste disposal and sanitation systems
Rondônia	1996	5	5	Conservation units
Amapá	1996	1.4	1.4	Conservation units
Rio Grande do Sul	1998	7	7	Conservation units
Mato Grosso	2001	7	5	Conservation units
			2	Waste disposal and sanitation systems
Mato Grosso do Sul	2001	5	5	Conservation units
Pernambuco	2001	6	1	Conservation units
			5	Waste management
Tocantins[a]	2002	13	3.5	Conservation units
			3.5	Solid waste disposal and sanitation systems; water protection
			2	Slash and burn control
			2	Local environmental policy
			2	Soil protection
Rio de Janeiro[b]	2007	2.5	1.1	Conservation units
			0.8	Water resources
			0.6	Solid waste management

Sources: Bernardes (1999), Grieg-Gran (2000), Loureiro (2002), CPRH (2003), SEPLAN (2003), Rio de Janeiro (2007), and Domingues (personal communication).

[a] Final implementation state in 2007.
[b] Final implementation state in 2011.

[3] A more detailed description is provided by Loureiro (2002, p. 79ff.) and May et al. (2002, p. 195).

Table 5.2 illustrates the types and shares of all ecological indicators adopted so far for ICMS-E allocation by the respective states. With regard to the waste management indicator in Minas Gerais, funds are allocated to those municipalities operating solid and liquid waste management systems duly licensed by the State Environmental Policy Council (Bernardes 1999).

Differences can be noted in the way the states introduced ecological indicators into the ICMS allocation system. In Rondônia, the allocation of ICMS based on value added (75 %), population (0.5 %), area (0.5 %) and agricultural production (5 %) did not change (Freitas 1999). It was the "equal share" indicator being reduced from 19 % to 14 % that allowed for the introduction of the ecological indicator, which was allocated a share of 5 % (Grieg-Gran 2000).

Whereas Rondônia changed its allocation system within one year (between 1996 and 1997), Minas Gerais introduced its new allocation system step-wise, starting to apply new indicators in 1996 and achieving full operation of the new system in 1998. In Minas Gerais, the introduction of the ICMS-E was part of a substantial change of the whole ICMS allocation system, popularly known as the "Robin Hood Law" (Bernardes 1999). The allocation based on value added was gradually reduced from about 94 % (1995) to 80 % (1998) to allow for the consideration of other indicators. Apart from the introduction of ecological indicators, eight further indicators were implemented: geographical area (1 %), population (2.71 %), 50 municipalities with the biggest population (2 %), education (2 %), area cultivated (1 %), cultural heritage (1 %), health expenditure (2 %) and the municipalities' own revenue generation (2 %). Here, the reform of the ICMS system covered not only ecological aspects, it also included the redistribution of resources for social reasons, aimed at making poorer municipalities better off. For this reason, it is much more difficult to illustrate clearly the effects of the introduction of ecological indicators in Minas Gerais compared to Rondônia (Grieg-Gran 2000).

The state of Tocantins also decided to introduce the ICMS-E in a gradual manner. Having started in 2003 with an overall percentage of 3.5 % for ecological indicators, Tocantins is to allocate a total of 13 % in 2007 for conservation units and indigenous areas, local environmental policy, slash and burn control, water and soil protection, as well as waste and sewage disposal (SEPLAN 2003).

A similar step-wise implementation is foreseen in the state of Rio de Janeiro, which passed its law on the ecological ICMS in October 2007 (Rio de Janeiro 2007). Starting with just 1 % in 2009, in its final implementation stage in the year 2011 a total of 2.5 % will be allocated to municipalities according to conservation units, the quality of water resources and solid waste management (see Table 5.2).

Before investigating the specific effects of the ICMS-E, a brief overview is presented regarding the existence and respective categories of protected areas for the states of Paraná, Rondônia and Minas Gerais. Table 5.3 shows the jurisdiction of protected areas in these states in 1997, differentiated according to federal, state and municipal levels. All states have in common a very low percentage of protected areas under municipal jurisdiction. In Paraná, more than 95 % of areas are protected under a jurisdiction higher than the municipal level; in Minas Gerais and Rondônia the figure is more than 99 %.

Rondônia represents a very special situation. This state has an extraordinarily high percentage of protected areas (36 %), with more than 50 % of all its municipalities containing protected areas (total number of municipalities in 1998: 52). Rondônia began to be settled through officially induced colonisation by people of European ancestry only in the late 20th century, starting in the early 1970s. A substantial part of this state consists of Indian reserves, extractive reserves, federal parks and other reserves, most of which are threatened by illegal logging, mining and other economic activities. In contrast, Paraná and Minas Gerais are in the Atlantic Forest, where settlement has taken place ever since colonisation in the 16th century, leading to substantial deforestation. Only 1 % of the original Atlantic Forest is left, of which 75 % is severely endangered. Protected areas in Minas Gerais account for 2 % of its total state area, with only about 16 % of its municipalities including protected areas as part of their municipal area (total number of municipalities in 1998: 853) (Grieg-Gran 2000). Paraná's protected areas also account for about 2 % of the total state area (total number of municipalities in 1999: 400).

Table 5.3: Jurisdiction of protected areas in Paraná, Rondônia and Minas Gerais (1997).

	Federal	State	Municipal	Total
Paraná				
Area protected (ha)	502,471	1,013,421	69,699	1,585,590
Percentage of total protected area	32	64	4	100
Protected areas as % of total state area				2
Minas Gerais				
Area protected (ha)	830,269	331,078	2,772	1,164,119
Percentage of total protected area	71	28	0.24	100
Protected areas as % of total state area				2
Rondônia				
Area protected (ha)	6,637,462	2,406,018	1,150	9,044,630
Percentage of total protected area	73	27	0.01	100
Protected areas as % of total state area				36

Source: Grieg-Gran (2000, p. 7).

5.3.3 Participation in the programme

In the states in which ICMS-E legislation exists, ecological fiscal transfers are automatically distributed to all municipalities that qualify, i.e. that hold officially registered conservation units within their territories. Therefore, an important prerequisite for the success of the programme is a good information policy. Municipalities need to know about the programme and they also need to expect benefits from participation for actually making an effort to create new conservation units or to use the ICMS-E revenues for additional conservation benefits. In

Minas Gerais, for example, the decentralised structure of the State Forest Institute, which is responsible for monitoring all information related to ICMS-E transfers based on conservation units, turned out to be extremely helpful in terms of publicising the new law. With its 150 local and 14 regional offices, it acted as an important source of information for municipalities with an interest in participation (Bernardes 1999).

May et al. (2002) indicate that since the programme's inception, Minas Gerais showed a 100 % increase in the number of municipalities benefiting from ICMS-E revenues. Currently, about 30 % of all municipalities in Minas Gerais participate in the programme. In Paraná, the ICMS-E programme began with the participation of 112 municipalities in 1992. In the year 2000, 221 municipalities benefited from the ICMS-E, either for conservation units or watershed protection or for having both types of protected areas. Over 50 % of all municipalities in Paraná are now participating in the programme. With regard to the biodiversity part of the ICMS-E programme, the number of participating municipalities grew from 63 in 1992 to 176 in the year 2000, representing an increase of 179 % (Loureiro 2002). This means that since its inception in Paraná, 113 municipalities qualified for the programme for the first time, due to the designation of new conservation units.

The ICMS-E has greatly improved relations between protected areas and the surrounding inhabitants (Bernardes 1999). Instead of perceiving protected areas as an obstacle to development, they are starting to see them as an opportunity to generate revenue. More municipalities are now aware of the existence of protected areas within their territory and are beginning to change their attitude towards them. They are more open to creating new reserves and, depending on the design of ecological indicators, also care about the quality of these areas. During the first year of operating the ICMS-E in Minas Gerais, only federal and state protected areas were considered; protected areas at municipal level were excluded because they lacked formal registration. The following year saw the official registration of existing municipal protected areas, and these were then included in the ICMS-E programme (Grieg-Gran 2000).

5.3.4 Increase in protected areas

Young (2005) stresses the effectiveness of the ICMS-E in encouraging the creation of new protected areas. Existing empirical studies concentrate on the states of Paraná and Minas Gerais, which were among the first states to introduce this economic instrument. Here, a clear incentive effect can be seen in the way new protected areas have been created, predominantly at local and state level. In relation to public protected areas in particular, the ICMS-E has become an important stimulus for the creation of new conservation units and for improved environmental management and quality of these areas.

In Paraná, the total area measured in conservation units grew by over 1,000,000 ha in the year 2000, representing an overall increase of 165 % during the 9 years since the programme's inception in 1992 (May et al. 2002). Table 5.4 shows that municipalities developed a strong interest in designating new public protected areas at the local level. The introduction of quality evaluation for conservation units had a positive effect on the interest of municipalities in improving their management (Grieg-Gran 2000, p. 21). Some municipalities and their mayors also started supporting private land users in managing conservation units, including provision of staff, equipment and vehicles for managing the areas.

In Minas Gerais, conservation units grew by slightly over 1,000,000 ha in 5 years, representing a 62 % increase (see Table 5.5, May et al. 2002). As for public protected areas, the designation of new protected areas was carried out exclusively at the state and municipal level of government. In Minas Gerais, however, the ICMS-E is not the only reason for the increase in protected areas: part of the initial growth was due to the efforts by local governments to register existing units that had not been regulated previously by the state.

Table 5.4: Public and private protected areas in Paraná before and after the ICMS-E.

Protected areas	Until 1991 (ha)	Created after 1991 (ha)	Total by 2000 (ha)	Increase (%)
Public				
Federal	289,582	50,846	340,428	18
State	39,859	13,804	53,663	35
Municipal	1,429	2,740	4,169	192
Private/mixed				
APA	306,693	905,631	1,212,324	295
RPPN	0	26,124	26,124	
Other	0	53,607	53,607	
Total	637,563	1,052,752	1,690,315	165

Source: Adapted from May et al. (2002). APAs (Environmental Protection Areas) can be designated at federal, state or municipal level. RPPNs (Private Natural Patrimony Reserves) can be designated at federal or state level.

Table 5.5: Public and private protected areas in Minas Gerais before and after the ICMS-E.

Protected areas	Until 1995 (ha)	Created after 1995 (ha)	Total by 2000 (ha)	Increase (%)
Public				
Federal	268,147	0	268,147	0
State	295,151	196,436	491,587	67
Municipal	3,851	9,076	12,927	236
Private/mixed				
APA	1,023,566	785,894	1,809,460	77
RPPN	20,261	13,808	34,069	68
Total	1,610,976	1,005,214	2,616,190	62

Source: Adapted from May et al. (2002). APAs (Environmental Protection Areas) can be designated at federal, state or municipal level. RPPNs (Private Natural Patrimony Reserves) can be designated at federal or state level.

In Paraná and Minas Gerais, intra-state allocations favour municipalities with large areas dedicated to 'indirect use' conservation units, which are designated at state and federal levels. Relatively high conservation weights are assigned to parks, reserves and forests (cf. Table 5.1), and a significant increase in protected areas can be noted for these management categories, at both state and municipal governmental levels (May et al. 2002). Nevertheless, a substantial volume of financial resources has been allocated to municipalities with Environmental Protection Areas (Área de Proteção Ambiental, APAs). APAs alone account for 86 % (Paraná) and 78 % (Minas Gerais) of the incremental increase in total new conservation units in these states, the vast majority being dedicated to state and municipal APAs. APAs are easily created, and a relatively low level of control is practised within them. They may cover large areas within a municipality with restricted zoning, in spite of far less rigorous enforcement than other conservation units, which is reflected by rather low conservation weights.

Following the introduction of the ICMS-E, new state legislation in Paraná and Minas Gerais has enabled the generation of a wider range of conservation units, including areas established under municipal jurisdiction as well as the Private Natural Patrimony Reserves (*Reserva Particular do Patrimônio Natural*, RPPN). RPPNs are owned and administered by a wide range of institutions, including NGOs, industry or private landowners. These private reserves are established and run by their owners. Owners are fully responsible for maintenance and management, yet they may benefit indirectly from the revenues municipalities receive based on the ICMS-E.

Originally, RPPNs represented a category of protected areas that was to be designated and handled at the national level by IBAMA (*Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis*), involving a very slow and difficult process. Paraná and Minas Gerais established state legislation in the 1990s, allowing a less bureaucratic way of designating and handling the private reserves at state level. Together with the financial incentive effect of the ICMS-E, this step contributed to a significant increase in the number and size of RPPNs. By the year 2000, some 26,124 ha were designated in Paraná and 34,069 ha in Minas Gerais. Both states are promoting RPPNs as part of an integrated public-private partnership in buffer zones surrounding public protected areas (Bernardes 1999). Particularly in Paraná, state environmental agencies play a very active role in promoting private reserves, and the state now holds the largest number and area of RPPNs (183 reserves with 36,928 ha) protecting the remaining Atlantic Forest, one of the global hotspots of biodiversity (Mesquita 2004). The positive experience with these reserves was triggered by the ecological fiscal transfers that motivated local governments to assist landowners in measures to protect and maintain the environmental quality of their areas, as well as helping them to prepare the necessary registration documentation (Bernardes 1999). A number of local governments have now begun to share additional revenues from RPPNs with landowners, whether in cash or in kind. This is done through agreements with the state association of RPPNs, since there is no leeway in the state laws for direct payment of municipal budgetary resources to private landowners for environmental services. May et al. (2002) mention in the case of Paraná that mainly large farmers were prioritised for RPPN creation, due to the size of their property and the volume of resources generated from it. Although small farmers may also be interested in RPPN creation, their land is not usually eligible due to its small size and the high transaction costs associated with it.

Bernardes (1999) presents figures for protected areas in Minas Gerais up to June 1999 that further break down the categories of APA and RPPN according to their governmental levels (Table 5.6).[4] These figures clearly show how strong an incentive effect the ICMS-E exercised on the initiative at state and local levels to create new protected areas. The largest absolute increase (552,976 ha) in protected areas up to June 1999 was due to new conservation units at state level. However, the largest relative increase of 1,619 % took place at municipal level, where 120,294 ha of protected areas existed in June 1999 compared to only 6,997 ha before the introduction of the ICMS-E.

Table 5.6: Protected areas in Minas Gerais before and after the ICMS-E, according to governmental levels.

Protected areas	Up to 1995 (ha)	Created after 1995 (ha)	Total by June 1999 (ha)	Increase (%)
Federal	909,467	5,710	915,177	0.6
State	695,610	552,976	1,248,586	79.5
Municipal	6,997	113,297	120,294	1,619.2
Total	1,612,074	671,983	2,284,057	41.7

Source: Adapted from Bernardes (1999). In this table, public and private/mixed protected areas, such as APAs and RPPNs, are already included in the figures presented according to their governmental level of designation.

5.3.5 Changing municipal revenues

The total amounts passed through to municipalities are appreciable. In the state of São Paulo, the sums distributed from the ICMS-E amounted to 70,241 million Brazilian Real (R$) between 1994 and 1996, averaging over R$ 23.4 million annually. It is estimated that participating municipalities in São Paulo received about R$ 2.45 per hectare of protected area per month (Bernardes 1999). Azzoni and Isai (1994) provide an estimate of the opportunity costs associated with protected areas for the state of São Paulo. They relate the value of production lost (in terms of state GDP) to state and county losses in terms of fiscal transfers and revenues from the value-added tax. They show for two extreme scenarios that – based on existing protected areas – the cost of environmental protection is likely to be between 0.05 % and 0.03 % of the state's GDP, depending on the location of the protected areas.

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[4] Bernardes (1999) and May et al. (2002) differ slightly in their base figure of protected areas in Minas Gerais up to 1995. Since the difference of less than 1 % is minimal, this is not considered any further here.

In Paraná, the amounts averaged over R$ 50 million annually between 1994 and 2000[5] (May et al. 2002). Individual municipalities, such as Piraquara in Paraná, saw their revenues increase considerably. Ninety percent of Piraquara's municipal territory is designated as a protected area for water conservation, while the remaining 10 % consists of protected areas for biodiversity conservation. This municipality increased its earnings by 84 % in 1995 (Loureiro, cited in Echavarría 2000).

May et al. (2002) investigated municipal revenues in the Paraná floodplain, the so-called "Varjão". Located in the northwestern part of the state, it lies within the Paraná, Paranapanema and Piquiri watersheds. Here, the ICMS-E constitutes a high percentage of overall municipal revenues and became a solution to the financial problems of the municipalities. The impacts of ICMS-E resources are especially significant, for example, in the municipality of São Jorge do Patrocínio, which has 52 % of its total area in conservation units (May et al. 2002, p. 185). In 1998, ICMS-E transfers represented 17.6 % of the overall municipal budget and in 2000, ICMS-E revenue amounted to 71 % of total ICMS transfers for that year. Conservation began to become an important part of the municipal agenda – for neighbouring municipalities as well. It led to the creation of Brazil's first municipal consortium for biodiversity protection in 1995 and, two years later, of the Ilha Grande National Park. Differences in ICMS-E pass-throughs are mainly due to the proportion of Park area within the total area of the municipality. The local population is aware of the financial importance of the ICMS-E for revenue generation, and the behaviour of the community towards the environment has changed. ICMS-E resources nowadays are used for numerous environmental activities, such as well drilling to provide drinking water, cleaning and landscaping of the urban area, rubbish collection, landfills, environmental education and the enforcement of land-use controls in parks and APAs. They are also used for other activities, such as the acquisition of tractors or the construction of industrial facilities. "All these benefits, provided by ICMS-E revenues, are disclosed to the community to make the public aware of the link between environmental protection and day-to-day-problems" (May et al. 2002, p. 185).

Grieg-Gran (2000) investigated the detailed financial effects of the ICMS-E for the states of Rondônia and Minas Gerais. Due to the relatively low number of municipalities in Rondônia, a full picture for the whole state is available. Here, the 5 % ecological share of total ICMS was introduced between 1996 and 1998 in combination with the equivalent reduction in the equal shares indicator. As a consequence, municipalities must have at least a 25 % share of protected area within their territory to outweigh the reduction in the equal shares indicator. Municipalities with more than a 25 % share of protected area can benefit significantly from the new type of transfers. In Rondônia, roughly 60 % of municipalities with protected areas benefited from the introduction of the ICMS-E: 7 municipalities with more than 50 % protected area, three municipalities with 40–50 %, six municipalities with 30–40 % protected area, and one municipality with 25–30 %. The remaining 40 % of municipalities experienced a negative impact due to the reduction in weighting accorded to the equal shares indicator.

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[5] In June 2001: Brazilian Real R$ 1.00 = US$ 0.41.

In Minas Gerais, the values allocated through the ICMS-E reached about R$ 15 million annually between 1998 and 2000 (May et al. 2002). In 1998, 86 municipalities with protected areas benefited from the increase in their consolidated index for ICMS allocation thanks to the ecological indicator (Grieg-Gran 2000). Thirty-eight municipalities with protected areas experienced a reduction in their overall index. This reduction in revenues resulted not so much from having too few protected areas, but rather from other factors. In part, it was due to the reduction in weighting accorded to value added; and in part it was to do with the introduction of other new indicators, such as health and education. In the case of Minas Gerais, it is important to distinguish between the effects of the ecological indicator and those of the others. For some municipalities, however, the introduction of the ecological indicator turned out to be extremely important. In 1998, the ecological index accounted for more than 20 % of the consolidated index in some 20 municipalities. The municipality of Marliéria, for example, has 55 % of its territory within the Rio Doce State Park, the largest contiguous area of Atlantic Forest in Minas Gerais. Here, the ecological index accounts for 70 % of the overall index for ICMS allocation (Grieg-Gran 2000), and in the first year after the ICMS-E was introduced, revenues from it accounted for around 68 % of total municipal ICMS revenues, increasing from R$ 36,648 in 1995 to R$ 811,335 in 1996 (Bernardes 1999).

One generic feature of the ICMS Ecológico in particular deserves to be highlighted: the amount of money available for distribution for ecological purposes in any one year depends on the state revenues of value-added tax in this year. Thus, municipalities are competing for the same pool of money. Given certain finite tax revenues, the creation of additional conservation units will inescapably lead to a gradual decrease in payments for environmental services per unit. Municipalities compete to provide more ecosystem services at lower and lower costs. Therefore, it is important to design the ICMS Ecológico not just around the quantity of conservation units, but also around their quality. In Paraná, the growing quality of conservation units over the years has demonstrated just how much of an incentive effect the criterion of quality is capable of generating (Loureiro 2002). If municipalities are aware of this feature, it may also motivate them to consider carefully the opportunity costs of different land uses.

Grieg-Gran (2000) presents calculations for Rondônia and Minas Gerais in order to identify the potential benefits that municipalities can expect to accrue when they set aside a further 1,000 ha of protected area.[6] For municipalities with very low average levels of value added and primary production, the conservation option would turn out to be highly beneficial. In the case of Rondônia, 28 municipalities would benefit from creating protected areas. Of these, only 12 already have protected areas, 16 have none and could be financially better off with protected areas. Comparative calculations for Minas Gerais are more complicated due to complex changes in overall ICMS-E allocations. Still, there are municipalities with low levels of value added where the creation of protected areas would be extremely financially attractive. São Sebastião do Rio Preto, which has no conservation units, would have to generate at

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[6] Grieg-Gran (2000) assumes that the municipality concerned is the only one in the whole state to designate an extra 1,000 ha of its territory as a protected area.

least 226 times the average value added per hectare in the municipality for it to be more beneficial in terms of ICMS revenues than the creation of 1,000 ha of protected area (Grieg-Gran 2000, p. 24).

5.4 The ecological ICMS: general reflections and transfer potential

5.4.1 The way forward in Brazil

Although the basic features of the ICMS-E are rather uniform across the various Brazilian states, the method of implementing it, its operation in practice and the reactions on the part of the municipalities can vary greatly. In-depth empirical studies show that ICMS-E allocations appear to have substantial impacts on conservation decisions in some areas, while in others only a limited impact can be observed (May et al. 2002). Municipalities with a high share of protected areas in particular can benefit substantially from the ecological fiscal transfers, and therefore appreciate the ecological services they provide across local boundaries. Many municipal governments – and depending on their information policy the public as well – are now aware of the natural assets they preserve and maintain. However, the type of indicator chosen also determines the incentive effect. The examples of Paraná and Minas Gerais show that not only the quantity but also the quality of the respective areas should be taken into consideration. In Paraná, there has not only been an increase in the number and surface area of conservation units, there have also been noted improvements in the quality of conservation units (Loureiro 2002). A quality-based evaluation awaits implementation in other states, and this represents a major challenge due to the regular controls of the registered areas conducted by decentralised environmental institutions. One prevalent point of critique relates to the way the ICMS-E revenues are currently allocated to municipalities. So far, they are given as lump-sum transfers, to be used in any way the recipient wishes. Some authors argue that earmarking for environmental purposes should be considered (Grieg-Gran 2000; May et al. 2002). However, from a general public finance perspective, there are also a number of arguments in favour of lump-sum transfers, such as guaranteeing maximum financial autonomy to local jurisdictions.

Despite the pros and cons of how precisely to design the ICMS-E, this innovative instrument is in the ascendent in Brazil. One of the great advantages of the ICMS-E is that it is not an instrument that requires new institutions or a new bureaucracy. By simply introducing an ecological indicator into the existing fiscal transfer mechanism, it builds on existing institutions and administrative procedures, thereby entailing very low transaction costs. The ICMS Ecológico is strongly promoted by the Federal Environmental Ministry and many initiatives have been undertaken by a variety of actors to advance its implementation in those states that do not have it so far, such as Goías, Espírito Santo and Bahía (Arantes 2006; Veiga Neto 2006, personal communication). A major challenge for Brazil will be to promote efforts for the design and implementation of economic incentives that are geared directly towards individual land users. The ICMS-E clearly has its focus on municipalities and on compensation for the land-use restrictions imposed on local public governments. However, a substantial proportion of the opportunity costs generated by protected areas accrues at the level of the individual land user. Therefore, further programmes need to be developed that focus on the individual land user, among them payments for environmental services, agricultural certification schemes, agri-environmental and forestry programmes, and so forth.

5.4.2 Transfer potential to other federal systems

Local governments rarely have substantial scope for influencing decisions made on the designation and maintenance of large-scale areas set aside for protective purposes. The "forced" provision of ecological goods and services in the form of protected areas, where there is no compensation for positive spillovers, is neither effective nor efficient. From an economic point of view, it is perfectly rational for local governments to be uninterested in – or even to be against – water and nature protection areas if the associated costs are to be borne locally, e.g. in terms of land-use restrictions, whereas many benefits cross local boundaries. Municipalities do not usually support the existence of protected areas within their territory, apart from a few exceptions where intrinsic motivation or a substantial potential for nature tourism comes into play. For most other local actors, protected areas reduce options for generating local income by attracting more inhabitants or promoting economic development. Even though protected areas might exist, a lack of enforcement, control or even simply information can easily lead to the deterioration of the quality of these areas. Therefore, one prerequisite for long-term sustainable land use consists in the integration of protected areas with positive spillovers into intergovernmental fiscal transfers to the local level. The internalisation of positive spatial externalities brings local interests in line with supra-local interests, thereby creating incentives for appropriate local public behaviour and contributing to economic efficiency.

One of the major problems associated with centralised provision of ecosystem services is that knowledge about the opportunity costs of protected areas is greater at the local level. The ICMS Ecológico addresses this problem by combining centralised incentives with decentralised decisions. As with markets, it does not require any centralised source of information. Although this article presented a national case study from Brazil, the general message can be transferred to other federal systems. It goes without saying that the recommendations to be made depend on the type of federal system being looked at, the general role and functions of different jurisdictions within these systems, and the specific constitutional and environmental legislation in force. In Brazil, the present focus is on compensating municipalities, i.e. public institutions, based on environmental performance indicators. Due to the specific legal framework and societal conditions prevailing there (including the prevention of corruption), there are almost no instruments that directly support private land users in their substantial role as conservation actors (Loureiro 2005, personal communication).

If we take the European Union and its many federally organised member states as a prominent example for comparison, the choice of instruments for compensating for local spillover benefits reflects the opposite strategy. Here, the focus is almost exclusively on the private land user, be it in agriculture, forestry or aquaculture. There are, for example, many different agri-environmental programmes in existence that are continuously being monitored and improved with the aim of increasing ecological effectiveness (e.g. Court of Auditors 2000; Baldock et al. 2002). Ultimately, both sides – private land users and the municipality as the public representative of local communities – have to be considered in their specific role for long-term sustainable land use (Ring 2004). However, in a few European countries there is a need for increased use of fiscal transfers to the local level for conservation purposes.

Köllner et al. (2002), for example, present a recent case study for integrating biodiversity into intergovernmental fiscal transfers in Switzerland. Portugal has recently set up a fiscal transfer scheme that explicitly rewards municipalities for having designated Natura 2000 sites and other protected areas within their territories (de Melo and Prates 2007). In Germany, this topic has already been discussed among the research community, federal conservation agencies and expert councils. In its reports over the last 10 years the German Advisory Council on the Environment has called for the integration of ecological indicators into intergovernmental fiscal transfers to the local level (SRU 1996; Ewers et al. 1997). A number of detailed studies and suggestions regarding implementation are available (e.g. Bergmann 1999; Rose 1999; Ring 2001; Perner and Thöne 2005), although the crucial step towards creating actual policy on this has not yet been taken. The Brazilian case showed that various mechanisms already exist for acknowledging ecological goods and services in intergovernmental fiscal transfers to the local level. In Germany, only very few states have implemented ecological aspects in their fiscal system, mostly concentrating on end-of-the-pipe and infrastructure-related public ecological functions (Ring 2002). The special relevance of protected areas has not yet been recognised. As a result, the majority of German municipalities still perceive them as an obstacle to development (Bauer et al. 1996, p. 334; Stoll-Kleemann 2001). Both theoretical analysis of the principles behind the economic theory of federalism related to spillovers from protected areas and the respective empirical investigations of the Brazilian federal system have shown that there still is a great need for adequately rewarding ecological services provided by the local level.

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Part III:

Applied biodiversity governance: Interdisciplinary and policy-relevant contributions

6 Biodiversity. Emerging issues for linking natural and social sciences

This chapter was published as “Jentsch, A., Wittmer, H., Jax, K., Ring, I., Henle, K. (2003). Biodiversity. Emerging issues for linking natural and social sciences. GAIA 12(2): 121–128”.

This is an interdisciplinary paper written by UFZ scholars working in the Department of Conservation Biology and the Department of Economics. It was prepared and published in the context of setting up a new biodiversity research plan within the Helmholtz Association’s research programme on “Sustainable Use of Landscapes” (2003–2008), aiming at integrative biodiversity research that is both scientifically sound and policy-relevant.

Abstract

More than a decade after the signing of the Convention on Biological Diversity, biodiversity loss is still occurring at unprecedented rates. There is a strong consensus that human activities are among the major drivers of biodiversity loss and that integrated approaches between natural and social scientists are needed. This paper develops the rationale for an integrative biodiversity research to produce socially relevant knowledge. It illustrates the challenges and potentials for integrative research using the examples of key drivers of biodiversity change: disturbances, invasions, and fragmentation as well as biodiversity conservation as a societal process. It concludes by highlighting conceptual changes necessary for integrative biodiversity research.

Keywords:

Biodiversity conservation, ecology, global change, interdisciplinary methodology, socio-economics

6.1 Introduction

Ten years after “Rio”, the research community has to ask itself whether its efforts – the research questions it is concerned with, the approaches it applies, the way it organizes, coordinates, and communicates its research – are suited to meet the growing challenge of biodiversity loss. It has convincingly been argued, including in notable contributions to this journal, that an interdisciplinary approach is needed to tackle environmental degradation. Indeed, research programs on biodiversity increasingly involve natural *and* social sciences disciplines, and funding organizations place high emphasis on interdisciplinary design in their programs. In spite of many efforts, linking social and natural sciences for integrative biodiversity research remains a major challenge. In this paper, we discuss new dimensions of this challenge. Our awareness results from the preparation of a biodiversity research plan within the new research program of the Helmholtz Association on "Sustainable Use of Landscapes". We argue that methods and tools to link natural and social sciences have to be adjusted to the specific issue under consideration in order to make biodiversity research both scientifically sound and policy-relevant.

6.2 Global change and loss of biodiversity

The current species-extinction period is mainly caused by human impact, and estimated to happen at a rate 1,000 times greater than the natural rate of extinction (Primack 1993). Recently, various Global-Change scenarios have been developed that address the effects on biodiversity caused by atmospheric warming, altered precipitation patterns, land-use changes, increased fragmentation, urban expansion, or other human activities (Sala et al. 2000). Ongoing discussions have been further alerted by the last report of the Intergovernmental Panel on Climate Change (IPCC 2001) stating an accelerated speed of change. The ability of living biota to adapt to such rapid environmental changes is questionable (Jentsch and Beierkuhnlein 2003; Kaule et al. 1999) (Figure 6.1).

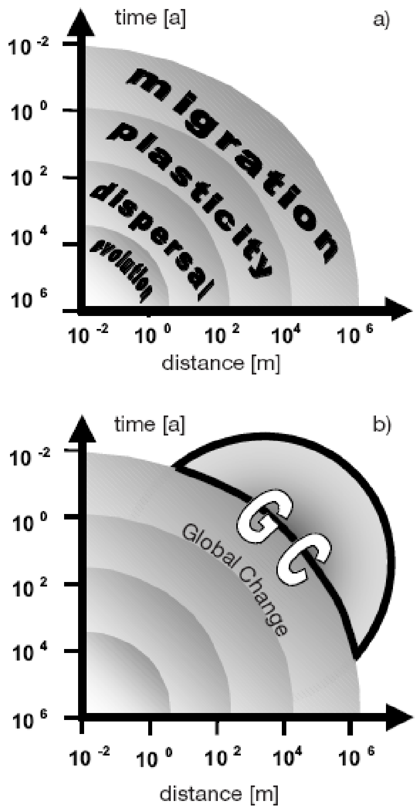

Figure 6.1: Spatio-temporal scales of biotic mechanisms to cope with changing environmental conditions: migration, plasticity, dispersal, and evolutionary development (a) versus the spatio-temporal scale of global change phenomena (b). As soon as species cannot cope with the speed or the spatial extent of environmental change, they are likely to go extinct (based on Beierkuhnlein 2003). Note the log/log scale.

The "crisis of biodiversity" discussed during the last two decades originated at first from a diffuse fear of losing species (Wilson 1985; Western 1992). In contrast to earlier mostly academic discussions on the relations between diversity and stability (Goodman 1975), the current debate on biological diversity was explicitly introduced by conservation biologists because of their concerns about the loss of biodiversity. We have understood that species diversity plays a vital role for the persistence of ecosystem processes and services in the face of severe land use pressure and changing environmental conditions (Hooper and Vitousek 1997). Meanwhile, the socio-economic value of ecosystem services is widely acknowledged (Costanza et al. 1997). Biodiversity, especially functional biodiversity, is increasingly recognized as decisive for maintaining these services (Hooper and Vitousek 1997). Still, it remains a fundamental challenge to assign economic and ethical attributes to particular species, communities or to ecological functions such as slope stabilisation or nutrient retention, in order to propose conservation measures where the obtained benefits exceed the costs for action. Likewise, it is necessary to better understand incentive structures in society that increasingly threat biodiversity and to develop mitigation options. Concepts for their implementation and for conflict reconciliation are required (Saunders et al. 1995).

Within the natural sciences, research approaches now extend from assessing regional species richness and characterizing the global patterns of biodiversity to analyzing functional implications of diversity, for, for example, biomass production, clean water, disease prevention, pharmaceutical use, or dynamic landscape equilibrium (Ernst et al. 2000; Loreau et al. 2002). Both the Convention on Biological Diversity[1] and current research programs such as the 6th framework program of the European Union explicitly call for linking natural and social sciences as essential focus when addressing the "drivers" for biodiversity dynamics and species loss. However, these postulates have rarely been implemented in a satisfactory manner.

6.3 Integrating natural and social sciences

Since its invention in 1986 in the preparation of the famous *Forum on Biodiversity* in Washington D.C. (Wilson 1988), "biodiversity" has been both a political and a scientific concept. This characteristic has experienced another strong boost through the development and ongoing implementation of the Convention on Biological Diversity. The primary causes of biodiversity loss are rooted in societal processes that lead to global change and the remedies to halt it depend on human values and on social and economic structures. The dynamics of these structures and the decision processes resulting from them have to be taken into perspective if biodiversity research is really to contribute to the protection and sustainable use of biodiversity (Brechin et al. 2002). At the same time, much ecological research is required to provide the basis for informed decision processes in this field. Disciplinary biodiversity research currently covers many aspects (compare boxes 6.1–6.4 for methodologies).

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1 http://www.cbd.int/convention/convention.shtml

Box 6.1: Biodiversity assessment

Biodiversity as defined in the Convention on Biological Diversity is not restricted to species diversity but encompasses the whole diversity of life, ranging from genes to ecosystems and biomes. These levels display complex spatial and temporal patterns and functional interactions. Therefore, biodiversity assessment needs to go beyond mere species-richness inventories. It has to account for spatial patterns, temporal rhythms, and anthropogenic processes impacting upon them. Global biodiversity patterns have recently been analyzed (Barthlott and Winiger 1998) and depicted, stimulating the discussion about biodiversity hot spots and global conservation priorities (Myers et al. 2000), an issue that has not yet been satisfactorily resolved and urgently requires the merging of biological and socio-economic/cultural research.

These aspects, that is, methods and knowledge from the natural and the social sciences, must be connected and integrated (Diversitas Scientific Committee 2002; WBGU 1996). The challenge is an applied, socially relevant biodiversity research to serve sustainable development.

As argued in the introduction, the research approaches in the natural and social sciences have to be linked in ways that are specific to the particular biodiversity-conservation challenge under consideration. To illustrate this point, we discuss in this section four challenges of biodiversity research and management addressed in the upcoming Helmholtz research program on “Sustainable Use of Landscapes”. We will pay specific attention to combinations of approaches that facilitate using research results in planning processes.

6.3.1 Disturbance, stability, and management options

Crucial drivers shaping the evolution of biodiversity are natural and anthropogenic disturbances such as floods, fire, bioturbation, or logging (Connell 1978; White and Jentsch 2001). Disturbances are ubiquitous, inherent, and unavoidable, causing both stability and change of ecological functions and distribution patterns (Turner et al. 1998; Jentsch et al. 2002). They can have huge ecological effects and economic costs, and their impact is difficult to mitigate (Figure 6.2). However, alteration of disturbance regimes can also be a threat to the maintenance of biotic diversity and may result in changes of the abundances of many species and of the composition of communities (Tilman 1996; Beierkuhnlein 1998). We lack simulation models on the implications of human activities on disturbance regimes at various spatial and temporal scales. We also lack an understanding of the vulnerability of ecosystems to disturbance and knowledge on the sensitive threshholds and preconditions at which disturbances become catastrophes.

Box 6.2: Experimental research

Biodiversity matters essentially to the performance of ecosystem functioning (Loreau et al. 2002). It contributes to productivity, can insure against environmental disasters, and provides numerous ecological services to mankind and human health (Heal 2000; Chivian 2001). Therefore, causal ecological analysis and experimental biodiversity research increasingly address functional interactions between species richness and ecological processes. Large-scale, sometimes continent-wide experimental settings have been developed to test the effect of species richness on, for example, biomass production in temperate grasslands (Hector et al. 1999) or productivity in agricultural environments. In the face of global change, results from first experimental manipulations predict high biodiversity to enhance ecosystem responses to elevated carbon dioxide and nitrogen deposition (Reich et al. 2001). Some important insights into ecological functions of biodiversity and driving factors for the decline of endangered species have been gained through the recent use of experimental designs on the landscape level (for example Caughley and Gunn 1996).

Figure 6.2: Annual flooding is a typical part of the disturbance regime of riparian ecosystems and maintains a high spatial and temporal niche diversity. However, the hazardous floods of Elbe and Mulde in Germany in August 2002 have shown, that extreme flooding events may have catastrophic effects and cause high economic costs. Managing floodplains for natural dynamics, biodiversity and human needs is a major challenge.

Box 6.3: Ecological modeling

Ecological modeling can be used as a unifying tool for integrating scientific results from various experimental and observational analyses as well as scenarios of changing environmental or socio-economic conditions (Drechsler and Burgman 2004). During the 1980s, grid-based[2] and individual-based[3] simulation models were established as new powerful tools to analyze ecological events in a spatially explicit framework (DeAngelis and Gross 1992; Grimm 1999). Existing grid-based models showed that spatio-temporal correlations imposed by, for example, disturbances are a key to understanding system dynamics, their vulnerability or resilience (stability). There is growing evidence that such correlations are the unifying currency to understand not only spatiotemporal dynamics of ecological systems, but also general mechanisms of the interaction between natural and anthropogenic drivers for biodiversity loss and species-specific functions and traits (Wiegand et al. 1999; Vos et al. 2001). Based on this knowledge, management strategies can be tested in order to find guidelines for biodiversity conservation, ecosystem restoration, and spatial planning (for example Henle et al. 1999). Recently developed ecological-economic models are promising techniques for further integration of social and natural sciences.

Even more than traditional approaches of biodiversity-conservation management, such as the establishment of protected areas or compensation schemes, management options including a promotion of disturbances require an integrated approach. Such strategies may include allowing events to take place that people have been trying to suppress for centuries such as fire and flooding. The efforts in terms of awareness creation and citizen deliberation necessary to achieve a consensus on such strategies are obviously higher than those needed for established conservation measures. Therefore, there is a need to assess different options and simulate their costs and benefits and the equity implications for different groups.

A promising tool that allows researchers to bring together knowledge from the natural and the social sciences and provides decision makers with a means to attach values to the implications of research results is Integrated Assessment (Horsch et al. 2001). It combines two tools for integration: ecological-economic modeling, and multicriteria decision analysis. Ecological-economic modeling is a pioneering approach for economic assessments of different ecological management options (for example Frank and Ring 1999; Johst et al. 2002). The modeling approach can establish the relationship between economic parameters of disturbance-management alternatives and the ecological effects. The possibility to use such integrated models to simulate different scenarios is a distinctive advantage of the approach.

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[2] Grid-based models subdivide an area into grid cells and model its fate by applying rules of change to each grid cell.

[3] In individual-based models the dynamics of a system are modeled by tracing the fate of individual objects (such as individuals of a population).

Box 6.4: Social sciences and biodiversity policies

From an economic perspective, an important task consists in determining monetary values of biodiversity that are not reflected in current market prices. These include (1) non-consumptive use values, such as the benefits species richness provides to tourism, (2) indirect use values for ecosystem stability or functions such as the provision of clean water, (3) option values such as future use in pharmaceuticals, (4) existence, and (5) bequest values (Costanza et al. 1997; Heal 2000; Perrings 1995; Fromm 2000; Dasgupta 2001). A debate has evolved that elaborates under which conditions economic valuation of biodiversity is sensible (Hampicke 1999; Nunes and van den Bergh 2001; Seidl and Gowdy 1999). Limiting factors include the nonsubstitutability of the natural resource, the fact that the societal preferences for the natural good to be valued cannot be represented by individual preferences, and institutional structures significantly influencing the results of the monetarisation.

A concept that has gained increasing importance is governance research, which has the objective to understand how decisions are made and what furthers their successful implementation (Kissling-Näf and Kern 2002). The analysis of governance structures focuses on the division of responsibilities between governmental levels as well as between public institutions, civil society and the private sector. With regard to the interaction between the different sectors, there is an extensive literature on participatory approaches and co-management of protected areas (compare for example Wells and Brandon 1992) and on relevant theoretical approaches from the social sciences concerning stakeholder involvement in applied biodiversity research (Wittmer and Birner 2001). A related branch of literature applies the potential of deliberative democracy for enhancing the social debate on biodiversity (compare for example O'Riordan and Stoll-Kleemann 2002). Research on global biodiversity governance further includes the investigation of incentive structures and policy instruments (Barbier 2000; OECD 1999). As biodiversity conservation typically causes costs at the local level while producing a global public good, special attention is paid to developing well-defined mechanisms for compensating local communities and land users (Ring 2002).

Psychology, sociology and cultural anthropology provide the methods for investigating individual and social perceptions of biodiversity (Escobar 1998). Biodiversity-related knowledge is increasingly being analyzed within the literature on local or indigenous knowledge and interfaces of different knowledge systems (Fairhead and Leach 1995; Long and Long 1992).

However, to decide on management strategies economic and ecological outcomes of the scenarios under consideration still need to be weighed against each other and against further criteria policy makers or other stakeholders consider relevant and that cannot be aggregated by means of a common denominator such as money. Multi-criteria decision analysis is an appropriate tool to solve this problem since it can rank alternatives using criteria with different units (Drechsler and Burgman 2004). Within Integrated Assessment stakeholders and policy makers define the management options to derive scenarios and attach the values to the different outcomes. Thus not only can results from different scientific disciplines be aggregated but also societal values are directly taken into account.

6.3.2 Invasion and precautionary principle

Biological invasions impose a major threat to biodiversity (for example Sala et al. 2000), have a great impact on ecosystem functions, and may cause substantial economic costs, for example in agriculture, forestry, tourism, hydraulic engineering, fisheries, and aquaculture (Mack et al. 2000). Basic challenges for the natural sciences are the systematic description of the stages of invasive processes (for example Heger 2001) and the identification of basic characteristics that distinguish those species having a high potential to become invasive and damaging in new environments from those who have not (Prinzing et al. 2002). Another important task is to understand and predict how species traits and the variables of the new habitat interact, that is, why some species become invasive in some ecosystems while they have insignificant effects in others (Byers et al. 2002).

Biological invasions are in most cases caused or exacerbated by human activities such as international trade, traffic or land-use changes (Figure 6.3). Comprehensive *ex post* calculations of the economic damage caused by invasive species have been published for selected countries (Pimentel et al. 2001). However, assessing the economic and ecological impact *ex ante* is necessary for decisions about control measures (Perrings et al. 2000). Biological impacts of invasions must be translated into terms of economic damage potential by cost-benefit analyses. The challenge consists in assessing the costs for combating invasions at different stages of the invasive process in order to determine cost-effective management strategies.

Preventing and managing biological invasions is also a challenge for legislation where the various governmental levels, from regional, national and European to the international level, have to be addressed. Acting in a very early stage of a potential invasion implies that precautionary measures are called for without sufficient scientific evidence that an invasion is about to take place. Such a procedure may be contrary to national, European, and other international rules and regulations. Therefore, the legal sciences must search for options to implement this principle in national and international environmental law in the presence of conflicting regulations. A key challenge currently addressed within our research program refers to the translation of the ecological and economic assessment of risks that are associated with biological invasions into corresponding legal instruments. To achieve this goal, together with our partners, we are currently developing an analytical process in which natural and social scientists jointly identify risk-mitigation options. These are evaluated for efficacy, feasibility, and impacts in order to recommend the most appropriate means to mitigate risks considered unacceptable. Specifically, we will analyze how interactions between species traits, cultivation, and socio-economic drivers of landscape heterogeneity contribute to the likelihood of establishment of invasive species across several European ecosystems (risk assessment), and the associated ecological and economic hazards. This analytical pathway will account for uncertainties due to the probabilistic nature of risk events.

Figure 6.3: The Giant Hogweed Rufen (Riesenbärenklau/Heracleum mantegazzianum) is an invasive species that typically invades along rivers or streets prone to recurrent disturbance. It has a very high growth rate, and outcompetes most other species. Combating the Giant Hogweed Rufen by mechanical means is only of limited success.

6.3.3 Fragmentation, spatial policies and conflict reconciliation

Landscape structures and biodiversity patterns are tightly linked. Landscape structures are increasingly modified by the accelerating land-use change (Figure 6.4) leading to the loss and fragmentation of habitats (Kaule et al. 1999). Fragmentation in turn is one of the major drivers of the current loss in biodiversity (Groombridge 1992). These processes are primarily driven by policy sectors such as agriculture, forestry, transportation, tourism, industrial/infrastructure development or fiscal policies.

Research efforts have considerably advanced our understanding of the interrelationships between landscape structures, fragmentation, and the viability of species (Henle et al. 2004). Likewise, various planning instruments to mitigate or reverse the effects of fragmentation have been developed and partly tested under applied conditions. Among these instruments are software (for example Frank et al. 2003) as well as generic principles and rules of thumb for assessing and mitigating the effects of fragmentation, and for planning habitat connectivity and revegetation programs (for example Henle et al. 1999; Freudenberger and Brooker 2004), or landscape indices (Jaeger 2002). Furthermore, there has also been considerable progress concerning our ability to identify particularly sensitive species and habitats for setting conservation priorities (Henle et al. 2004). Thus, the development of principles for designing representative networks of conservation areas (Margules and Pressey 2000) is rapidly advancing.

Figure 6.4: Fragmentation of ecological units, such as the Araucaria angustifolia forests in southern Brazil, will cause local extinction of species as soon as they cannot overcome the distance between two units of their habitat by dispersal or migration mechanisms. Spatial planning therefore needs to carefully reconcile conflicts between land-use developments and biodiversity-related goals.

In spite of the progress in understanding the ecology of fragmentation our success in mitigating, halting, or reversing the effects of fragmentation has been limited. One reason for this is the fact that the different disciplines within natural science as well as of natural and socio-economic sciences have not yet been integrated sufficiently in the field of fragmentation. Most attempts to forecast the effects of habitat loss and fragmentation on biodiversity are based on a static view of the landscape or use simple scenarios without considering their different likeliness. Conversely, analyses of drivers of fragmentation, especially transport, infrastructure, and agricultural policy in Europe and forestry and land-use policies in tropical countries rarely consider the spatial effects of these policies on landscape structures and fragmentation on a scale that allows linking with existing knowledge on the ecological effects of fragmentation. Therefore, we have only a limited understanding why the same types of drivers lead to different dynamics of landscape structures in different regions. For example, why has the clearing of vegetation lead to a variegated landscape structure in Eastern Australia but to a pattern of isolated remnants with sharp edges in Western Australia (Freudenberger and Brooker 2004)? Likewise, our ability to predict, intentionally select among, and manage potential future trajectories of landscapes and landscape structures based on the analyses of societal choices and constraints is limited except for the short-term local effects of planning projects.

Understanding and mitigating the effects beyond the local level requires the integration of biodiversity concerns in a wide array of policy sectors. For example, strategies need to be devised that effectively maintain large unfragmented landscapes for the conservation of species with high area requirements but at the same time take into account the needs of people. We are currently analyzing how, within German fiscal policies, a region can be compensated for options in economic development it foregoes by maintaining unfragmented landscapes and whether, and if so how, the compensation should be linked to the relative importance of a region in terms of biodiversity-conservation strategies. *Inter alia* this requires to determine what decisions should be made at what level of government in order to manage fragmentation.

It has become clear (Frank and Ring 1999; Freudenberger and Brooker 2004) that the optimal solutions asked for by biologists analyzing the consequences of fragmentation are rarely feasible within socio-economic constraints and that the chances of success are low if biologists only pass on the results of their analyses to managers hoping that they would implement one of their preferred options. Therefore, we have developed ecological-economic modeling approaches based on the premise that they have to find acceptable compromises for the balance of ecological and economic requirements (Frank and Ring 1999; Johst et al. 2002). To help resolve emerging goal conflicts we have integrated models that allow analyzing the viability of species in the face of habitat fragmentation into multi-criteria decision analyses (Drechsler and Burgman 2004). The merging of these two conceptual approaches holds strong promises for resolving goal conflicts when integrating biodiversity concerns into other policy sectors.

GIS-based spatial modeling is a further promising tool to facilitate the co-operation across disciplines and with decision makers when dealing with habitat fragmentation. It integrates data from different disciplines in a spatial framework and thereby allows researchers to identify changing landscape patterns and their causes. By visualizing potential effects the models also allow decision makers to become aware of the spatial implications of different policies (McKelvey et al. 1993). However, besides such tools intensive communication among the disciplines and with stakeholders is essential to jointly identify the problems and develop solutions.

6.3.4 Biodiversity conservation and stakeholder participation

The concept of "biodiversity conservation" differs from the traditional measures of nature conservation that focussed on separating nature from humans. The Ecosystem Approach, which was approved by the 5th Conference of the Parties to the Convention on Biological Diversity (http://www.cbd.int/ecosystem/) states in its first principle that "the objectives of management of land, water and living resources are a matter of societal choice". Research on biodiversity can be framed in a socially relevant way if it links conservation with the question of sustainable use and benefit sharing, as implied by the Convention on Biological Diversity. The challenge for the implementation consists in re-conceptualizing biodiversity conservation as a social process (Figure 6.5). As such, this challenge differs from the challenges discussed above – disturbance, fragmentation and invasion – that are mainly defined by natural-science research.

Figure 6.5: Biodiversity conservation as a social process: non-governmental organizations in Thailand collecting 50,000 signatures in order to present a community forestry law to congress.

A starting point for understanding biodiversity conservation as a social process is to consider that biodiversity is affected by human resource use, either by the utilization of the species concerned or by using the area these species require. Human resource use is based on the established governance structures, the rules, norms and traditions in a society. In most cases these governance structures must be modified in order to achieve biodiversity conservation. This, however, implies addressing essential questions related to economics, ethics, social justice, and political governance, such as: Who benefits? How and by whom are decisions made and what is perceived as legitimate by those affected? Who is accountable, and to what extent are the parties involved complying with their obligations? Brechin and co-workers (Brechin et al. 2002) argue for a more explicit consideration of social-justice issues in biodiversity conservation. This consideration could produce more legitimate results and would thereby offer stronger foundations for compliance as well as for fair enforcement.

The challenge for biodiversity research consists in informing the societal actors who influence biodiversity in such a way that they feel this information is useful for them. This implies that biodiversity conservation will most effectively occur if the relevant actors, whether they are politicians or farmers, feel biodiversity conservation is an issue of interest to them. Therefore, research needs to acquire a deeper understanding of how societal rules about resource use are agreed upon and modified. Taking this perspective of the affected actors often implies posing research questions differently. An example could be: How to achieve sustainable use of certain species in a manner considered legitimate by the stakeholders? Combining scientific knowledge from different disciplines with local knowledge becomes necessary.

A major challenge for ecology in order to make the research relevant to stakeholders is to translate scientific concepts, especially that of the ecosystem, into a language which can be understood by and is useful to non-scientists, including indigenous communities. The Ecosystem Concept offers a strong potential as a tool for structuring interdisciplinary research projects. This approach makes use of the fact that perceptions and conceptualizations of the ecosystem and of biodiversity are also socially determined and strongly depend on the interests of different stakeholders. The ambiguity of the Ecosystem Concept, sometimes perceived as a weakness can here be turned into a strength by allowing to compare and integrate different social views of nature and its (socially valued) components (Jax et al. 1998; Jax 2002). In return, such an approach will help to refine ecological theory and the direction of empirical studies, by testing the usefulness and compatibility of different ecosystem concepts. Research can thus be focused on those of the possible entities that matter to the perception of nature by people while at the same time allowing for useful predictions of ecological dynamics.

6.4 Perspectives

The above examples of challenges for integrative biodiversity research illustrate the necessity to move away from the traditional division of labour between the sciences. The traditional model is characterized by the view that natural science research should provide an assessment and understanding of biodiversity and derive strategies for biodiversity conservation in order to pass these on to the social sciences. Social sciences should then calculate the value of biodiversity, assess the costs and benefits and the social acceptability, and translate the conservation strategies into instruments to be presented to policy makers. This approach, however, does not take into account how and why societal decisions that negatively affect biodiversity are made. If biodiversity change is reconceptualized as an ecological, social and political process mutual exchange is required at much earlier stages.

Integrating findings from different disciplines, scales, and dimensions beyond a mere additive approach requires special tools. Instruments with a promising potential include, among others, ecological-economic modeling, spatial modeling, participatory multi-criteria decision analysis, and Integrated Assessment. As the examples of disturbance, invasion and fragmentation have shown, the specific needs for integration differ considerably.

The discussion of biodiversity conservation and sustainable use shows that a new perspective is necessary if biodiversity loss and conservation are seen in the context of social processes: the point of view of the actors affecting and affected by biodiversity change. This requires a much closer collaboration and interaction among sciences and between science and society. As a result, it has the potential for deriving management options that are feasible and can actually be implemented in a given context. In turn, this kind of research poses new questions to theory. The experiences in integrating local and scientific knowledge and promoting stakeholder participation in biodiversity management contribute to achieving this reconceptualisation in biodiversity research. This development will be greatly enhanced if researchers receive training in interdisciplinary communication (compare Box 6.5) and if incentives such as high profile prices, career advantages, and priority funding for groups that bridge the social and natural sciences in an innovative way can be provided.

Box 6.5: Interdisciplinary learning cabinets

This method has been designed by one of the authors (H. Wittmer) for the course program of the Interdisciplinary Graduate College on "Valuation and Management of Biodiversity" at the University of Göttingen, Germany. Young researchers (doctoral students) who came from different disciplines (ecology, economics, social sciences, and law) presented the approaches and methodologies of their respective disciplines to the other students. The mission was to use practical examples so that the fellow students could experience themselves what it means to be a researcher from the respective discipline: What questions are raised, what perspectives are taken, what methods used? For example, the ecologist took the others on a field trip and let them identify species, calculate diversity indices and develop an argumentation strategy for or against renaturalizing a section of a river from the standpoint of biodiversity conservation. In law a court on a case of international fisheries law was enacted and participants had to find out whether the construction of a windmill was permissible with respect to nature-conservation legislation.

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7 Naturschutz in der föderalen Aufgabenteilung: Zur Notwendigkeit einer Bundeskompetenz aus ökonomischer Perspektive

Nature conservation competencies in federal systems –
The economic arguments for national-level competencies

This chapter was published as “Ring, I. (2004). Naturschutz in der föderalen Aufgabenteilung: Zur Notwendigkeit einer Bundeskompetenz aus ökonomischer Perspektive. Natur und Landschaft 79(11): 494–500”. The German journal “*Natur und Landschaft*”, although not included in journal citation indexes, is nonetheless highly significant due to being the main conservation journal read by decision-makers and administrators at all governmental levels.

In October 2003, a Commission of the German Parliament (*Bundestag*) and the Federal Council (*Bundesrat*) was inaugurated to modernise the German federal system, i.e. the intergovernmental relations between federal and state level (*Föderalismuskommission*). In the course of the discussions that followed, the German Federal Agency for Nature Conservation organised an expert workshop on “Nature conservation in the federal system” (*Naturschutz im föderalen System*, 24 March 2004, Bonn, Germany). I. Ring was invited as an expert to present an economic perspective on the issue, giving a talk entitled: “Nature conservation in the federal system – federal or state public function? The economic perspective” (*Naturschutz im föderalen System – zentrale oder dezentrale Aufgabe? Die ökonomische Perspektive*). She was among the few selected speakers who were subsequently asked to publish her talk in “Natur und Landschaft” in order to make the economic perspective available to a wider readership.

7.1 Einführung

Im Oktober 2003 haben Bundestag und Bundesrat die Kommission zur Modernisierung der bundesstaatlichen Ordnung, kurz: Bundesstaatskommission, eingerichtet, die Vorschläge zur Neuverteilung der Gesetzgebungskompetenzen zwischen Bund und Ländern erarbeiten soll. Zu den Zielen der Kommission gehört es, die Handlungs- und Entscheidungsfähigkeit von Bund und Ländern zu verbessern, die politischen Verantwortlichkeiten besser zuzuordnen sowie die Zweckmäßigkeit und Effizienz der Aufgabenerfüllung zu steigern. Dabei sollen die Verteilung der Gesetzgebungskompetenzen auf Bund und Länder, die Zuständigkeiten und Mitwirkungsrechte der Länder bei der Gesetzgebung des Bundes sowie die Finanzbeziehungen zwischen Bund und Ländern überprüft werden. Darüber hinaus sind sowohl Aspekte der europäischen Integration als auch die Situation der Kommunen zu berücksichtigen.

Die Arbeit der Kommission baut u.a. auf den Beschlüssen der Ministerpräsidentenkonferenz vom 27. März 2003 auf, die die Abschaffung der Rahmenkompetenz des Bundes gefordert hatte, verbunden mit einer Stärkung der Ländergesetzgebung[1]. Im Verlauf der bisherigen Arbeit der Bundesstaatskommission haben einige der berufenen Sachverständigen in ihren Stellungnahmen die Abschaffung der Rahmengesetzgebung für Naturschutz, Wasserhaushalt und z.T. weitere Sachtitel im Umweltrecht mit einer überwiegenden Verlagerung in die Zuständigkeit der Länder gefordert (Huber 2003; Schmidt-Jortzig 2003; Scholz 2003; Benz 2003). Dem entgegen positionieren sich andere Sachverständige hinsichtlich einer Befürwortung der Bundeskompetenz für das gesamte Umweltrecht (Meyer 2004) oder argumentieren differenziert hinsichtlich der zu berücksichtigenden externen Effekte und Skalenerträge im Umweltbereich (Scharpf 2004).

Weitgehend Konsens herrscht in der Kommission hinsichtlich der Abschaffung der derzeitigen Kategorie der Rahmenkompetenz des Bundes (Zypries 2003). Hält dieser Konsens an, wird dies auch Konsequenzen für die Bereiche Naturschutz und Wasserhaushalt haben. Hier ergeben sich zwei Alternativen: Einerseits bietet sich die Überführung in die konkurrierende Gesetzgebung des Bundes an, was eine integrierte Behandlung des Naturschutz- und Umweltrechts auf Bundesebene erlauben würde. So hat der Umweltausschuss des Deutschen Bundestags parteiübergreifend eine Lösung gefordert, die es erlaubt, ein Umweltgesetzbuch auf Bundesebene zu erlassen. Für eine Stärkung des Naturschutzrechts auf Bundesebene plädieren die Umweltverbände (Röscheisen 2004), namhafte Rechtswissenschaftler (Koch und Mechel 2004) sowie der Rat von Sachverständigen für Umweltfragen in seinem aktuellen Umweltgutachten (SRU 2004, Tz. 224). Andererseits könnte die Bundesstaatskommission auf Grund ihrer bis Ende 2004 angelegten Tätigkeit zu dem Ergebnis kommen, Naturschutz- und/oder Wasserrecht – z.T. sogar weiter gehende Umweltkompetenzen, wie kürzlich von den Präsidenten der Landtage von Schleswig-Holstein, Bayern, Rheinland-Pfalz, Thüringen und Sachsen-Anhalt vorgeschlagen (Arens et al. 2004) – auf die Länderebene zu verlagern und damit empfehlen, die Bundeskompetenz auf diesen Gebieten aufzugeben.

In der Diskussion um die Neuordnung der föderalen Aufgabenteilung werden neben politischen und rechtlichen vielfach ökonomische Argumente hinsichtlich der Vor- und Nachteile der Zentralisierung bzw. Dezentralisierung von staatlichen Aufgaben vorgebracht. Allerdings finden die ökonomischen Argumente momentan nur relativ pauschal und plakativ Eingang in die gegenwärtige Diskussion. So beziehen sich Befürworter der Verlagerung von Kompetenzen auf die Länderebene häufig auf die Förderung eines föderalen Wettbewerbs; Gegner der Dezentralisierung von Naturschutz- und Umweltkompetenzen führen meist die Befürchtung eines „ruinösen“ Deregulierungswettbewerbs bzw. „Race-to-the-bottom“ an. Aus ökonomischer Sicht können diese pauschalen Positionen nicht befriedigen, da sie in der Regel ohne eine nachvollziehbare Begründung vorgebracht werden. Ziel des vorliegenden Beitrags ist es, ökonomische Argumente für bzw. gegen Naturschutzkompetenzen auf Bundesebene differen-

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[1] Einen Überblick und eine Einschätzung zu den aktuellen Vorschlägen und neuen Formen der Gesetzgebungskompetenzen aus umweltrechtlicher Perspektive geben Koch und Mechel (2004).

ziert und sachgerecht zu diskutieren. Auf der Grundlage der ökonomischen Theorie des Föderalismus werden relevante Kriterien vorgestellt, die eher für eine Zentralisierung oder Dezentralisierung von Aufgaben des Natur- und Umweltschutzes sprechen. Für die Ableitung konkreter Empfehlungen hinsichtlich der Kompetenzzuweisung im Naturschutz ist schließlich die räumliche Verteilung seiner Nutzen und Kosten zu berücksichtigen.

7.2 Naturschutz: Dezentrale oder zentrale Aufgabe?

Jede konkrete Umsetzung von Naturschutzpolitik in föderalen Systemen muss die spezifischen ökologischen Eigenschaften des jeweiligen Guts berücksichtigen (Ring 2002). Im Folgenden werden einschlägige ökonomische Kriterien für die Bereitstellung eines öffentlichen Guts vorgestellt, die für die Verteilung der Gesetzgebungskompetenzen im Naturschutz herangezogen werden können.

Prinzipiell sind die Raumwirksamkeit der jeweiligen Aufgaben und die Mobilität der betroffenen Umweltmedien oder Schutzgüter zu berücksichtigen. So sprechen großräumige öffentliche Aufgaben wie die Bereitstellung von Schutzgütern von nationaler oder internationaler Bedeutung für eine zentrale Aufgabenwahrnehmung. Das Gleiche gilt tendenziell für grenzüberschreitende Phänomene. Letztere treten bei hochgradig mobilen Umweltkompartimenten oder Schutzgütern auf, welche eine Abstimmung der diesbezüglichen Aufgaben auf nationaler, oft sogar internationaler Ebene bedürfen (Siebert 1991; Hansjürgens 1996). So gibt es zahlreiche Tierarten, wie etwa große Wirbeltiere und Zugvögel, deren großräumiger Habitatanspruch oder Wanderungsverhalten administrative Grenzen überschreiten und entweder eine koordinierte dezentrale oder eine nationale, mitunter auch internationale Schutzpolitik nötig machen (Abbildung 7.1). Auch die großräumige Vernetzung von Habitaten im Naturschutz erfordert eine zentrale Koordination, wie etwa der Aufbau des europäischen Schutzgebietssystems Natura 2000.

Öffentliche Aufgaben mit kleinräumiger und regional begrenzter Wirksamkeit, wie die Bereitstellung lokaler öffentlicher Güter, lassen sich auf dezentraler Ebene effektiv und effizient gestalten (Ring 2002). Ebenso sind Aufgaben im Zusammenhang mit weniger mobilen Umweltkompartimenten verstärkt dezentralen Gebietskörperschaften zuzuordnen. Dies hängt mit der geringeren Wahrscheinlichkeit grenzüberschreitender Umweltbelastung und damit der Existenz räumlicher Externalitäten zusammen. Fragen der Landnutzung und Bodenbelastung, aber auch öffentliche Aufgaben in Verbindung mit Binnengewässern können gewöhnlich innerhalb nationaler Grenzen gelöst werden. Das Gleiche gilt für viele Teilaspekte der Durchführung des Naturschutzes und der Landschaftspflege. In Abhängigkeit von den spezifischen Eigenschaften der Schutzgüter bzw. Umweltprobleme lassen sich auch Ansätze für eine weiter gehende Dezentralisierung auf die Ebene der Länder oder kommunalen Gebietskörperschaften begründen.

Foto: D. Nette

Abbildung 7.1: Bedrohte Tierarten mit großem Habitatanspruch wie der Luchs (*Lynx lynx*) benötigen national und international gesetzte Naturschutzstandards sowie eine länderübergreifende Koordination der Artenschutzaktivitäten.

Figure 7.1: Endangered species with large habitat requirements such as the lynx (*Lynx lynx*) need national and international standard-setting and coordinated conservation activities.

Aus ökonomischer Sicht gilt bezüglich der Bereitstellung öffentlicher Güter, zu denen der Naturschutz zählt, das Prinzip der fiskalischen Dezentralisierung (Oates 1972). Danach lässt sich die Bereitstellung der öffentlichen Güter und Dienstleistungen am effizientesten gestalten, wenn sie der niedrigst möglichen Gebietskörperschaft zugewiesen wird.

Eine Abweichung von dieser generellen Dezentralisierungsregel ergibt sich bei der Existenz räumlicher Externalitäten (Spillover) zwischen Gebietskörperschaften. Im Naturschutz gibt es solche positiven grenzüberschreitenden Effekte für Schutzgüter von überregionalem Interesse, z.B. für Schutzgebietskategorien mit internationaler Bedeutung (Nationalparks, FFH-Gebiete, Biosphärenreservate). Auch negative grenzüberschreitende Effekte existieren bei Eingriffen in Natur und Landschaft mit großräumigen Auswirkungen (z.B. Infrastrukturvorhaben mit grenzüberschreitenden Wirkungen). Das Prinzip der fiskalischen Äquivalenz gebietet jedoch, eine Übereinstimmung zwischen denjenigen herbeizuführen, die über den Umfang des anzubietenden Kollektivguts entscheiden, die den Nutzen aus einem kollektiven Gut ziehen und jenen, die für die Bereitstellung dieses Guts zahlen müssen (Olson 1969). Die gesellschaftliche Wohlfahrt wird durch die differenzierte Ausgestaltung öffentlicher Leistungen entsprechend der räumlichen Verteilung von Kosten und Nutzen erhöht. Die Verwirklichung des Prinzips der fiskalischen Äquivalenz erfordert aber nicht notwendigerweise die Verlagerung von Kompetenzen auf übergeordnete Gebietskörperschaften. Verhandlungen zwischen den betroffenen regionalen Einheiten oder die Bildung neuer administrativer Einheiten, die die räumliche Ausdehnung der Entscheider, Kosten- und Nutzenträger abbilden, können auch zur Internalisierung räumlicher Externalitäten herangezogen werden (z.B. regionale Kooperation in Stadt-Umland-Verbänden oder Zweckverbände).

Im folgenden Kapitel wird der Bereich des Naturschutzes einer differenzierten Betrachtungsweise unterzogen und geprüft, inwieweit aus ökonomischer Sicht, d.h. bezüglich der räumlichen Verteilung seiner Nutzen und Kosten, eine eher dezentrale oder zentrale Bereitstellung zu befürworten ist.

7.3 Die räumliche Verteilung von Nutzen und Kosten des Naturschutzes

7.3.1 Die räumliche Verteilung der Nutzen des Naturschutzes

Der Gesamtwert des Guts Naturschutz ergibt sich in der umweltökonomischen Theorie aus der Summe seiner Teilwerte. Die verschiedenen Nutzenstiftungen des Naturschutzes werden in der Ökonomie aus einer anthropozentrischen Sichtweise betrachtet; dabei geht es um die Wertschätzung auf der Basis individueller Präferenzen von Menschen (Hampicke 1991, S. 107ff.; Pearce und Turner 1990, S. 141ff.). Dies schließt nicht aus, dass neben quantifizierbaren Werten, die eine Monetarisierung erlauben, auch qualitative Elemente, die schwer oder nur unter großer Unsicherheit in Geldwerten auszudrücken sind, in die Bewertung eingehen.

Prinzipiell wird bei der ökonomischen Bewertung von Naturressourcen zwischen nutzungsabhängigen und nicht nutzungsabhängigen Werten unterschieden (vgl. Abbildung 7.2). Die Nutzungswerte werden in direkte und indirekte Nutzungswerte unterteilt, wobei zu den direkten Nutzungswerten der Produktions-, der Konsum- und der Symbolwert des Naturschutzes gehören (WBGU 1999, S. 54ff.). Beispiele für den biosphärischen Produktionswert sind die Produktion und Vermarktung von Holz, Getreide, Baumwolle oder Fischen. In der Regel handelt es sich hier um private, marktgängige Güter, wobei die Rahmenbedingungen für eine nachhaltige Nutzung der Naturressourcen durch staatlich gesetzte Regeln zu gewährleisten sind, damit es nicht zu einer Übernutzung kommt. Zum Konsumwert des Naturschutzes zählen der Erlebniswert, Freizeit- und Erholungswerte sowie der ästhetische Wert von Natur und Landschaften. Der Konsumwert zeichnet sich vorwiegend durch Kollektivgutcharakter aus. Der Symbolwert fasst weitgehend religiöse und spirituelle Werte von Natur (z.B. heilige Tier- und Pflanzenarten) zusammen, die von Individuen bestimmten Elementen der Biosphäre zugewiesen werden.

Indirekte Nutzungswerte ergeben sich aus den Funktionen und biosphärischen Leistungen der Ökosysteme. Hierher gehören beispielsweise zahlreiche Funktionen biogeochemischer Kreisläufe, das Hochwasserretentionsvermögen von Auenlandschaften oder die Rolle des Bodens als Filter für Schadstoffe. Oft werden diese Funktionen erst bei ihrem Verlust wahrgenommen. Da von ihrer Nutzung niemand ausgeschlossen werden kann, handelt es sich bei deren Bereitstellung um öffentliche Aufgaben.

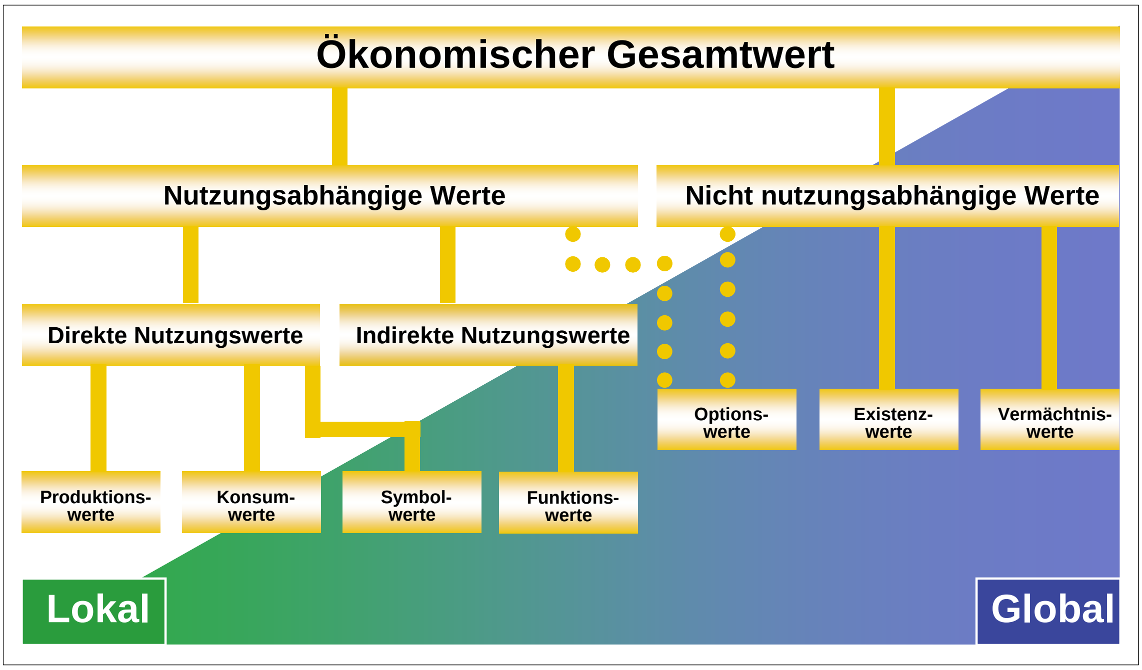


Abbildung 7.2: Der Nutzen des Naturschutzes in seiner räumlichen Ausprägung.

Figure 7.2: The economic benefit of nature conservation and its spatial relevance.

Zur Kategorie der nicht nutzungsabhängigen Werte zählt vor allem der Existenzwert. Bei ihm handelt es sich um Nutzenstiftungen aus dem Wissen um die Existenz von Objekten, seien dies bedrohte Tier- und Pflanzenarten oder außergewöhnliche und seltene Landschaften (z.B. Weltnaturerbe). Der Existenzwert wird im Allgemeinen unabhängig von jeglicher gegenwärtiger oder zukünftiger Nutzung formuliert. Des Weiteren fällt der Vermächtniswert in die Kategorie der nicht nutzungsabhängigen Werte, der aus dem Wunsch resultiert, Elemente der Biosphäre auf Grund ihrer Symbol- und Identifikationswerte an nachkommende Generationen zu vererben.

Optionswerte beziehen sich auf künftige, potentielle Nutzenstiftungen des Naturschutzes und können sich sowohl auf nutzungsabhängige als auch auf nicht nutzungsabhängige Werte beziehen. Zu den nutzungsabhängigen Optionswerten gehören etwa Optionen der künftigen Nutzung des Genpotenzials für medizinische Zwecke, deren konkreter Nutzen heute aber noch unbekannt ist.

Um Empfehlungen hinsichtlich der Zentralisierung oder Dezentralisierung von Naturschutzkompetenzen abzuleiten, sind die räumlichen Verteilungen der Nutzenstiftungen des Naturschutzes zu berücksichtigen (vgl. auch Döring 1998). Generell lässt sich sagen, dass Naturschutzaktivitäten sowohl von lokalem als auch globalem Nutzen sein können (Perrings und Gadgil 2003). Der räumliche Nutzenkreis nimmt aber im Allgemeinen von den direkten über die indirekten Nutzungswerte hin zu den Options- und Existenzwerten zu (Abbildung 7.2. Allerdings können auch die Produktionswerte schon von überregionaler Bedeutung sein, je nach Ausmaß und Intensität von Landnutzungen bzw. Eingriffen in Natur und Landschaft. Darüber hinaus bestehen ökologische Wechselwirkungen von Eingriffen vor Ort mit überregionalen, nationalen oder internationalen Schutzzielen, so dass es gerade im Naturschutz einer Koordination von Maßnahmen auf einer höheren Ebene bedarf.

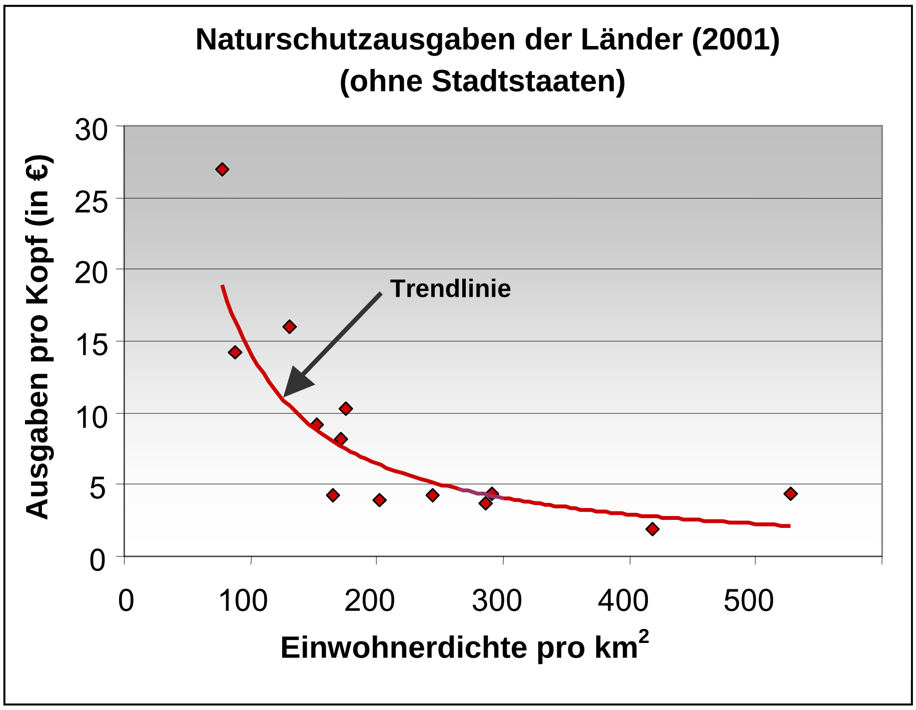
Aus ökonomischer Sicht wird als unstrittig eingeschätzt, dass insbesondere die nicht nutzungsabhängigen Werte natürlicher Ressourcen als Rechtfertigung einer Bundeskompetenz heranzuziehen sind (Oates 2001; Revesz 2000). Der Artenschutz und der Schutz der biologischen Vielfalt sind damit originäre Aufgaben zentraler Gebietskörperschaften. Dezentrale Gebietskörperschaften verhalten sich mit hoher Wahrscheinlichkeit als Trittbrettfahrer, wenn die Kosten von Maßnahmen für den Artenschutz und den Schutz der biologischen Vielfalt selbst zu tragen sind, während ihr Nutzen teilweise oder überwiegend außerhalb der eigenen Gebietsgrenzen anfällt. Diese Tendenz wird auch durch empirische Studien gestützt: Eine Studie von List et al. (2002) hat in den USA die öffentlichen Ausgaben für bedrohte Arten (nach dem *Endangered Species Act*) auf der Ebene der Bundesstaaten im Verhältnis zur Bundesebene untersucht. Dabei zeigte sich, dass die Umweltbehörden der einzelnen Staaten dazu tendieren, weniger auszugeben, wenn es um den Schutz von Tierarten mit großem Habitatanspruch geht oder wenn Konflikte von Artenschutzzielen mit wirtschaftlicher Entwicklung im eigenen Wirkungskreis zu erwarten sind.

Das Argument von einem „ruinösen“ Deregulierungswettbewerb ist im Naturschutz also nicht unbegründet und spricht für eine zentrale Standardsetzung. Gleichzeitig bedarf die zentrale Entscheidungskompetenz einer zentralen Finanzierungskompetenz. Überlässt man die Finanzierung ausschließlich den dezentralen Gebietskörperschaften, so ist auf Grund des grenzüberschreitenden Nutzens des Naturschutzes mit einer Unterfinanzierung und damit einem Unterangebot der entsprechenden Schutzgüter zu rechnen. Dies gilt auch, wenn die Durchführung der entsprechenden Aufgaben auf der Ebene der Länder angesiedelt ist.

7.3.2 Die räumliche Verteilung der Kosten des Naturschutzes

Naturschutz ist ein öffentliches Gut, deshalb hat der Staat mit entsprechenden Rahmenbedingungen dafür zu sorgen, dass dieses Gut öffentlich bereitgestellt wird. Damit verbunden ist die Übernahme der Kosten des Naturschutzes durch die öffentliche Hand. Aufgrund des überregionalen Nutzens des Naturschutzes ist dieser eine gesamtgesellschaftliche Aufgabe, insofern sind die Kosten – folgt man dem Prinzip der fiskalischen Äquivalenz – auch von der ganzen Gesellschaft zu tragen. In Realität verhält es sich jedoch so, dass die Kosten des Naturschutzes in Deutschland ungleichmäßig verteilt sind.

Abbildung 7.3 veranschaulicht die Pro-Kopf-Ausgaben der Länder für den Naturschutz im Jahre 2001 und setzt diese zu der jeweiligen Einwohnerdichte ins Verhältnis. Dabei zeigt sich, dass die Pro-Kopf-Ausgaben der Länder ansteigen, je geringer diese besiedelt sind. Die Länder mit geringer Einwohnerdichte tragen also in überproportionaler Weise die Kosten des Naturschutzes in Deutschland. Dies hängt damit zusammen, dass Naturschutz in aller Regel „in der Fläche“ stattfindet.



Quelle: BfN, 1.1 (2004); Daten: Stratmann (2002). / Source: BfN, 1.1 (2004); Data: Stratmann (2002).

Abbildung 7.3: Naturschutzausgaben der Länder im Verhältnis zu ihrer Einwohnerdichte (2001).

Figure 7.3: Public conservation expenditures at state level in relation to population density (2001).

Von außerordentlicher Bedeutung für den Naturschutz sind die Land- und Forstwirtschaft. Einerseits waren diese beiden Wirtschaftssektoren auf Grund ihrer Produktivitätssteigerung und zunehmenden Intensivierung der Landnutzung in überdurchschnittlichem Maße am Artenrückgang des letzten Jahrhunderts beteiligt (Hampicke 1991, S. 45). Andererseits betreffen die heute für den Naturschutz anfallenden Kosten vor allem diesen Bereich. Denn extensive Landnutzungsformen, die gleichzeitig dem Natur- und Landschaftsschutz förderlich wären, sind unter gegenwärtigen wirtschaftlichen Rahmenbedingungen in der Regel nicht wettbewerbsfähig. Ohne eine breite gesellschaftliche Unterstützung in Form der Honorierung ökologischer Leistungen der Land- und Forstwirtschaft lässt sich der Rückgang der biologischen Vielfalt daher nicht aufhalten. Eine Honorierung ökologischer Leistungen ist aus ökonomischer Perspektive gerechtfertigt, da der Nutzen naturschutzgerechter Landnutzungsformen im Sinne des Erhalts der Biodiversität weit über den betroffenen Sektor hinaus reicht (Hampicke 2005). Die Flächenstaaten mit ihrem hohen Anteil an Land- und Forstwirtschaft sind hier überproportional betroffen, so dass sich die entsprechenden Kosten des Naturschutzes und der Landschaftspflege nicht gleichmäßig auf alle Länder verteilen.

Die ungleichmäßige Verteilung der Kosten des Naturschutzes betrifft auch die räumliche Verteilung der Schutzgebiete in Deutschland und die damit verbundenen Kosten. Sind die Landschaftsschutzgebiete in Deutschland noch relativ homogen verteilt, so zeigt sich ein Gradient der zunehmenden räumlichen Konzentration von den Naturschutzgebieten über die Biosphärenreservate, Naturparks und Nationalparks (Urfei 2002). So liegen nahezu 90 % der deutschen Nationalparkfläche in Mecklenburg-Vorpommern, Niedersachsen und Schleswig-Holstein (Abbildung 7.4).

Foto: Naturfoto-Pretscher

Abbildung 7.4: Der Nationalpark Jasmund auf Rügen, Mecklenburg-Vorpommern. Allein die drei Bundesländer Mecklenburg-Vorpommern, Niedersachsen und Schleswig-Holstein stellen nahezu 90 % der bundesdeutschen Nationalparkfläche (Urfei 2002).

Figure 7.4: The Jasmund National Park on Rügen in Mecklenburg-Western Pomerania. Almost 90 % of German national park area is located in the three states of Mecklenburg-Western Pomerania, Lower Saxony and Schleswig-Holstein (Urfei 2002).

Die Kosten der Ausweisung und des Unterhalts, die mit großräumigen und hochwertigen Schutzgebietskategorien zunehmen, sind in der Bundesrepublik Deutschland nahezu vollständig durch die jeweiligen Länder zu tragen[2]. Hierzu zählen beispielsweise die Einrichtung eigenständiger Schutzgebietsverwaltungen für Nationalparks und Biosphärenreservate. Gleichzeitig sind mit der Ausweisung von Großschutzgebieten lokale Nutzungseinschränkungen – und damit Opportunitätskosten – verbunden; diese betreffen sowohl die Land-, Forst- und Teichwirtschaft als auch die Gemeinden vor Ort. Die Gemeinden werden in ihren Entwicklungsmöglichkeiten bezüglich Siedlungs- und Gewerbenutzung eingeschränkt und können

[2] Die Bundesebene fördert z.B. Naturschutzgroßprojekte mit gesamtstaatlich repräsentativer Bedeutung (BfN 2002, S. 165). Für den regelmäßigen Unterhalt von Großschutzgebieten ist in Deutschland die Länderebene zuständig. Hier liegt der Unterschied zu Österreich. Obwohl es dort kein Naturschutzgesetz auf Bundesebene gibt, beteiligt sich die Bundesebene zu 50 % an den Kosten für die österreichischen Nationalparks (Loiskandl 2004, persönliche Mitteilung).

ihre kommunale Hoheit bezüglich der Flächennutzung nur noch eingeschränkt ausüben. Die privaten und öffentlichen Nutzungseinschränkungen führen dazu, dass sich häufig große lokale Widerstände gegen die Einrichtung beispielsweise neuer Nationalparks formieren. Aus ökonomischer Perspektive wäre hier ein Kostenausgleich in Form von Kompensationszahlungen für private Landnutzer, aber auch für lokale Gebietskörperschaften über den kommunalen Finanzausgleich zu fordern (Ring 2001, 2002; Perner und Thöne 2002). Ökonomische Anreize allein können allerdings den Widerstand gegen die Ausweisung hochwertiger Großschutzgebiete nicht brechen, hier bedarf es zusätzlich differenzierter Studien bezüglich ihrer Wahrnehmung und Akzeptanz (Stoll-Kleemann 2001). Aus ökonomischer Sicht lässt sich zusammenfassen: Je inhomogener die räumliche Verteilung in Verbindung mit der nationalen oder gar internationalen Bedeutung einer Schutzgebietskategorie, desto höher ist der zentrale Koordinierungsbedarf, d.h. die Notwendigkeit einer übergeordneten Bundeskompetenz.

Der Naturschutz hat nicht nur damit zu kämpfen, dass seine Kosten räumlich ungleichmäßig verteilt sind. Problematisch für das öffentliche Gut Naturschutz ist zusätzlich seine Unterfinanzierung (BfN 2002, S. 174; SRU 2002, S. 25). So betrugen die öffentlichen Naturschutzausgaben (Bund und Länder, ohne Stadtstaaten und Kommunen) im Jahr 1998 nur 1,22 Mrd. DM oder 0,067 % der Gesamtausgaben öffentlicher Haushalte. Im Vergleich dazu wurden für den Umweltschutz im selben Jahr 21,7 Mrd. DM an öffentlichen Ausgaben getätigt (Stratmann 2002). Die Unterfinanzierung hat zur Folge, dass z.B. Nationalparkverwaltungen ihren Aufgaben nicht nachkommen können, Defizite in der Ausbildung von Naturschutzfachkräften auftreten (Stoll-Kleemann 2001), eine zu dünne Personalausstattung in Naturschutzbehörden vorliegt (SRU 2002, S. 25) und nicht zuletzt unzureichende Lastenausgleiche auf der Seite öffentlicher als auch privater Landnutzer zu beklagen sind. So führt nicht nur die ungleiche räumliche Verteilung der Kosten und Nutzen, sondern auch die zu geringe öffentliche Mittelbereitstellung für den Naturschutz zu einem Unterangebot an dem nötigen Ausmaß und der Qualität von Schutzgütern.

7.4 Föderale Kompetenzverteilung auf Grund der Nutzen und Kosten des Naturschutzes

Naturschutz ist ein öffentliches Gut, dessen Nutzen überwiegend von nationaler und internationaler Bedeutung ist. Die Kosten des Naturschutzes sind sowohl sektoral (Land- und Forstwirtschaft) als auch räumlich (Schutzgebiete) ungleichmäßig verteilt. Dieser Zusammenhang erfordert eine übergeordnete, d.h. nationale oder gar internationale Rahmen- und Standardsetzung im Naturschutz. Insofern ist die fortschreitende und anspruchsvolle Standardsetzung auf europäischer Ebene, z.B. durch die FFH-Richtlinie und den Aufbau des Natura-2000-Netzwerks, aus ökonomischer Sicht zielführend. Ebenso ist die zunehmende europäische Ko-Finanzierung der Agrarumweltprogramme der Länder, auch im Hinblick auf Naturschutzziele, im Sinn der ökonomischen Föderalismustheorie sinnvoll zu begründen (vgl. Osterburg und Stratmann 2002). Nun wird von Länderseite häufig argumentiert, dass zur Umsetzung der europäischen Standards im Naturschutz keine gesonderte nationale Naturschutzgesetzgebung nötig sei, denn die Umsetzung könne direkt durch die Ländernaturschutzgesetzgebung erfolgen. Dieser Haltung wird von zahlreichen Autoren die mangelnde Europatauglichkeit der deutschen Umwelt- bzw. Naturschutzgesetzgebung entgegengesetzt (stellvertretend Pernice 2004; Hendrischke 2004). Für die fristgemäße und qualitativ gute Umsetzung der EU-Richtlinien ist die Bundesebene verantwortlich, unabhängig von der internen, nationalen Regelung der Zuständigkeiten, die im Naturschutzrecht eine entsprechende Anpassung von 16 Ländernaturschutzgesetzen erfordert. Aus ökonomischer Sicht vernachlässigt die Argumentationslinie der Länder auch die Rolle des föderalen Wettbewerbs, der auf allen föderalen Ebenen stattfindet. So setzt das Bundesnaturschutzgesetz für die Länder verbindliche Standards, die über europäische Vorgaben hinausgehen. Hier sind die flächendeckende Landschaftsplanung, die Durchsetzung des Verursacherprinzips über die Eingriffsregelung und der Entwicklungsgedanke im Naturschutz, der über das europäisch vorgegebene Verschlechterungsgebot von Flächen hinausgeht, zu nennen. Einerseits nimmt Deutschland damit in diesen Bereichen – z.T. mit einigen anderen europäischen Staaten – eine Vorreiterrolle im europäischen Staatenverbund ein. Andererseits setzt die Bundesebene damit flächendeckend für das Bundesgebiet Standards, die in ihrem Anspruch gefährdet wären, wenn es keine Bundeskompetenz für den Naturschutz mehr gäbe.

Die bislang angeführte, ökonomisch begründete Argumentation spricht für die Beibehaltung bzw. sogar Stärkung der gegenwärtigen Entscheidungskompetenz des Bundes im Naturschutz. Insofern ist die Übernahme des Naturschutztitels in die konkurrierende Gesetzgebung ökonomisch zu unterstützen, was den Weg für eine integrierte Behandlung des Umwelt- und Naturschutzrechtes auf Bundesebene bereiten könnte. Verbunden mit der Entscheidungskompetenz wäre allerdings auch die Finanzierungskompetenz des Bundes zu stärken. Die gegenwärtige Unterfinanzierung des Naturschutzes führt zu einem Unterangebot an entsprechenden Schutzgütern, was durch die ungleiche räumliche und sektorale Verteilung der Kosten des Naturschutzes noch verstärkt wird. Neben der Bereitstellung ausreichender Mittel auf allen föderalen Ebenen entsprechend dem räumlichen Nutzen der Naturschutzmaßnahmen sind ökologische Lastenausgleiche zu befürworten. Diese sollten Kompensationszahlungen sowohl für öffentliche Gebietskörperschaften im Rahmen des Länder- und kommunalen Finanzausgleichs als auch für private Landnutzer durch die Honorierung ökologischer Leistungen einschließen.

Danksagung

Der Artikel folgt in wesentlichen Teilen einem Vortrag, den die Autorin im Rahmen des Fachgespräches „Naturschutz im föderalen System“ am Bundesamt für Naturschutz (24.03.2004) gehalten hat. Für kritische Anmerkungen und Ergänzungen möchte ich mich insbesondere bei Prof. Bernd Hansjürgens sowie Prof. Wolfgang Köck bedanken.

Zusammenfassung

Durch die Arbeit der Kommission zur Modernisierung der bundesstaatlichen Ordnung ist die derzeitige Ausgestaltung der Rahmengesetzgebung in Frage gestellt worden, deren Veränderung zu weit reichenden Konsequenzen für die Naturschutzgesetzgebung führen könnte. Der vorliegende Artikel untersucht die ökonomischen Argumente für bzw. gegen eine Ansiedlung von Naturschutzkompetenzen auf den verschiedenen föderalen Ebenen.

Naturschutz ist ein öffentliches Gut, dessen Nutzen überwiegend von nationaler und internationaler Bedeutung ist. Die Kosten des Naturschutzes sind sowohl sektoral (Land- und Forstwirtschaft) als auch räumlich (Schutzgebiete) ungleichmäßig über die Länder verteilt. Dieser Zusammenhang erfordert eine nationale Rahmen- und Standardsetzung im Naturschutz. Aus ökonomischer Sicht ist deshalb eine Beibehaltung bzw. Stärkung der gegenwärtigen Entscheidungskompetenz des Bundes im Naturschutz zu fordern.

Summary

The Commission of the German Parliament and Bundesrat on the Modernisation of the Federal Structure has questioned the present design of framework regulation in Germany. Since nature conservation law in Germany is subject to framework regulation at the federal level, combined with implementing conservation laws at the state level (Länder), a potential change in framework regulation might have serious consequences for conservation law. This article investigates the economic arguments for and against conservation competencies at various governmental levels.

Nature conservation is a public good with benefits accruing predominantly at national and international levels. The costs of conservation are distributed unequally – in terms of economic sectors (agriculture and forestry) and in terms of the spatial distribution of protected areas across the various German Länder. These findings suggest a need for national-level competencies and standard setting. The economic perspective supports keeping or even strengthening nature conservation competencies at national level.

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8 Biodiversity governance: Adjusting local costs and global benefits

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8.1 Linking biodiversity governance to fiscal federalism

8.1.1 Multilevel and multiactor governance in biodiversity conservation

The concept of governance itself is associated with a wider perspective on environmental decision making.[1] Regarding the public side with mostly hierarchical decision making, it is no longer the nationstate alone that is predominantly in charge of setting the framework for environmental decisions. Both higher and lower levels of environmental decision making come increasingly into play, for example, European directives have had a strong hierarchical impact on member states regarding the creation of the Natura 2000 network; the international Convention on Biological Diversity, although more reliant on consensual decision making, still morally binds and influences signatory states regarding its implementation. State, regional and local governments can play increasingly crucial roles for environmental decision making. On the one hand, this depends on the type of federal system in place and its associated hierarchical rules. On the other hand, new alliances emerge in the form of hybrid organizations, such as Local Agenda 21 initiatives and networks. At the same time, environmental decisions are no longer the exclusive realm of governmental agencies. The private sector – including consumers and business, as well as hybrid organizations crossing the public–private divide such as NGOs or semiprivate organizations (for example, agencies and research institutes) – play an important role in decisions on the conservation and sustainable use of biodiversity.[2]

Hence, biodiversity governance has to be seen in a context of both multilevel and multiactor governance. Another important issue is the development and implementation of policy instruments. Regulation, economic incentives, voluntary, informative and communicative

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1 Biodiversity governance is understood as a subfield of resource governance. Environmental governance is used as the most generic term in this chapter and includes both of these fields. Concrete examples are mostly related to the conservation and sustainable use of biodiversity.

2 See Penker (2008) for a more detailed portrayal of private, public and hybrid actors related to landscape governance that is also valid for the field of biodiversity governance.

instruments on the public side as well as activities initiated by civil society continuously interact and are of varying importance depending upon the concrete problem. Regulation or so-called command and control instruments have been widely used in the early phases of environmental policy, representing the coercive principle in public state management of environmental goods. Nowadays, economic incentives and voluntary instruments are in the forerun wherever feasible, allowing individuals more leeway of action and contributing to increased economic efficiency. While they are policy instruments – meaning framed and issued mainly by state action – they no longer fit the perspective of the coercive state. Jordan et al. (2003) have framed the term 'new environmental policy instruments' with respect to the increasing use of these new instruments of environmental governance. The mix of instruments and thus the increasing interaction of public and private spheres, which is the larger topic of this book (see Sikor et al. 2008), also apply to the field of biodiversity governance.

One of the main objectives of this chapter is to link biodiversity governance to the economic theory of fiscal federalism. The latter is concerned with the assignment of public functions, expenditures and fiscal instruments to different levels of government. Therefore, we can identify an important link between multilevel governance and fiscal federalism where spatial characteristics of biodiversity conservation are of prime importance. The environmental quality of a landscape is closely linked to its land use pattern and the type of management performed by public jurisdictions and private land users. Yet, there are few incentives for local actors to encourage conservation activities when ecological benefits cross local boundaries (Perrings and Gadgil 2003). Spatial externalities or spillovers exist that – if not adequately compensated – lead to an underprovision of the public goods and services concerned. This is the case for a number of ecological services, such as nature reserves or the conservation of endangered wildlife. Decisions on the designation of protected areas or species protection are often made by institutions above the local level, whereas the concrete consequences in terms of restrictions in land use or the damages caused by wildlife are born by local actors, often without any or sufficient compensation.

Fiscal federalism as a subfield of public finance in economics predominantly deals with the public aspect of environmental governance, implying identification with the kind of goods and services that should be provided by public institutions at different levels of government. In this context, the distinction between public and private clearly refers to the state as the public sphere and the market as the private sphere of voluntary interaction (see Sikor et al. 2008). Fiscal federalism already involves a truly multilevel perspective on the provision of public goods and services; analysis is not confined to the nationstate as predominant public actor. Public institutions may range from international institutions down to municipalities as the most decentralized governmental level. It is important to note the difference between the municipality – as the local authority representing the state and the local community – and the wider political community including further local associations and stakeholders. Fiscal federalism deals with the local level of government. In the first part of this chapter, two important principles in fiscal federalism are introduced – the principle of decentralization and the principle of fiscal equivalence. They are applied to environmental issues, helping to decide on adequate levels of actions and relevant policy instruments.

The second part of this chapter analyses the spatial characteristics of conservation benefits and costs in relation to governmental levels. The costs and benefits of providing public goods and services do not always match with jurisdictional boundaries, which call for adequate instruments to adjust them. It is important to check who is bearing the costs and who is benefiting from certain conservation programmes or measures. In practice, public and private actors continuously interact and commonly contribute to the final outcome in terms of successes and failures of the conservation and sustainable use of biodiversity. Therefore, one has to consider public and private actors in the course of adjusting the benefits and costs of biodiversity conservation.

The third part of the chapter is devoted to a case study, presenting an innovative instrument implemented in the 1990s in several Brazilian states: the ICMS-Ecológico (Imposto sobre Circulação de Mercadorias e Serviços-Ecológico) (Grieg-Gran 2000; Loureiro 2001; May et al. 2002; Ring 2004b). It is an excellent and rare example of the consideration of ecological indicators in intergovernmental fiscal transfers from the state to the local level. The ICMS-Ecológico (ICMS-E) renders protected areas within municipal borders into communal revenues. In this way, it clearly represents an incentive for the creation of newly protected areas, mainly at local and state levels. Although a hierarchical, state-governed instrument, it is an economic instrument by nature. Furthermore, the ICMS-E emerged as an economic incentive for new forms of public–private interaction. Thus, the instrument and its effects are a good example of the shifting boundaries between notions of publics and privates in various aspects.

8.1.2 Environmental federalism

A central objective of fiscal federalism is to effectively and efficiently assign public functions, expenditures and revenues to the central, state and local governmental levels in federal systems. This means deciding which public functions and instruments are best centralized and which are better placed at decentralized levels of government (Oates 1999). Concerning the allocation function of public sectors, which is of major interest for the provision of environmental goods and services, the basic principle of fiscal decentralization has been put forward (Musgrave 1959; Oates 1972). The provision of most public goods and services is more efficiently guaranteed when production and consumption are limited to the lowest governmental level possible. The decentralization rule is based on the following major arguments. First, the regionally differing preferences can be better considered at decentralized levels of government (Tiebout 1956). Second, it is a requirement for more competition within the public sector. And third, many public goods and services experience cost degression due to lower coordination and transaction costs at the local level.

Nevertheless, there are exceptions from the general decentralization rule, as it only applies in the absence of economies of scale. If economies of scale exist, the provision of public goods and services concerned should be moved to the cost efficient centralized level (Postlep and Döring 1996). In addition, due to the characteristics of nonrivalry and nonexcludability of many public goods and services, some of them are associated with spatial externalities or spillovers between jurisdictions. Concerning environmental 'bads', such as pollution, the activities of one jurisdiction can lead to negative externalities in neighbouring or even remote jurisdictions, implying that part of the costs in question are externalized. In the context of biodiversity conservation, for example, building infrastructure such as roads may cause a fragmentation of habitats for protected endangered species, eventually leading to a decline or extinction of a population. Concerning environmental goods or services, spillover benefits may be created. In this case, a jurisdiction bears the costs of providing certain environmental goods and services, but the benefits thereof also accrue to people in other jurisdictions at the same or at higher governmental levels. For example, various types of nature reserves, such as national parks or biosphere reserves, lead to costs in terms of land use restrictions for local communities but are highly valued, even on a global scale.

When spatial externalities exist, the principle of fiscal equivalence comes into play. It advocates achieving a match between those who decide on the provision of a collective good, those who receive the benefits, and those who pay for it (Olson 1969). Social welfare is increased through the differentiation of public goods and services in accordance with spatially related costs and preferences. There are several ways of solving this problem. The implementation of fiscal equivalence may require the shifting of competence to a more centralized level of government. Other options involve negotiations between the parties concerned for more equal distribution of the costs and benefits of actions. This may include the formation of administrative institutions that map the spatial range of costs and benefits to internalize spatial externalities. In the EU, the implementation of the Water Framework Directive is currently leading to the establishment of new river basin authorities, or alternatively to cooperative arrangements among those jurisdictions that share a river basin, allowing for coordinated water management at the basin level (Petry and Dombrowsky 2007). Yet another option relates to fiscal instruments such as transfers or compensation payments that help to internalize spatial externalities.

What are the basic consequences of the decentralization rule and the principle of fiscal equivalence for environmental issues? Following the general decentralization rule for the allocation function of public services, the provision of environmental goods and services should be assigned to lower levels of government where appropriate. But what does this exactly mean in terms of environmental policy in general and biodiversity governance in particular? Not all problems can be dealt with at the local level. What could the criteria be for deciding on the suitability of a problem's decentralization?

In an initial approximation, the spatial scale of a problem and the mobility of the respective environmental media play an important role in deciding on the appropriate governmental level (Ring 2002). Consequently, small-scale public functions such as purely local public goods or land use related tasks are often suitable to local governmental decision making. The provision of drinking water or sewage and waste disposal are good examples of public services predominantly provided at the municipal level. By contrast, land related environmental problems accumulating at larger spatial scales, such as nitrogen emissions in agriculture, definitely need centrally set standards yet require local action. Spatial scale and mobility are also very important characteristics regarding species protection. The larger the habitat, for example, of large vertebrates, and the more mobile a species is (consider migrating species!), the greater the relevance of centrally performed services, including standards of protection. Concrete implementation of species protection policies still requires substantial local action, but this needs to be coordinated between public and private actors depending on the spatial distribution and mobility of the species concerned (Similä et al. 2006).

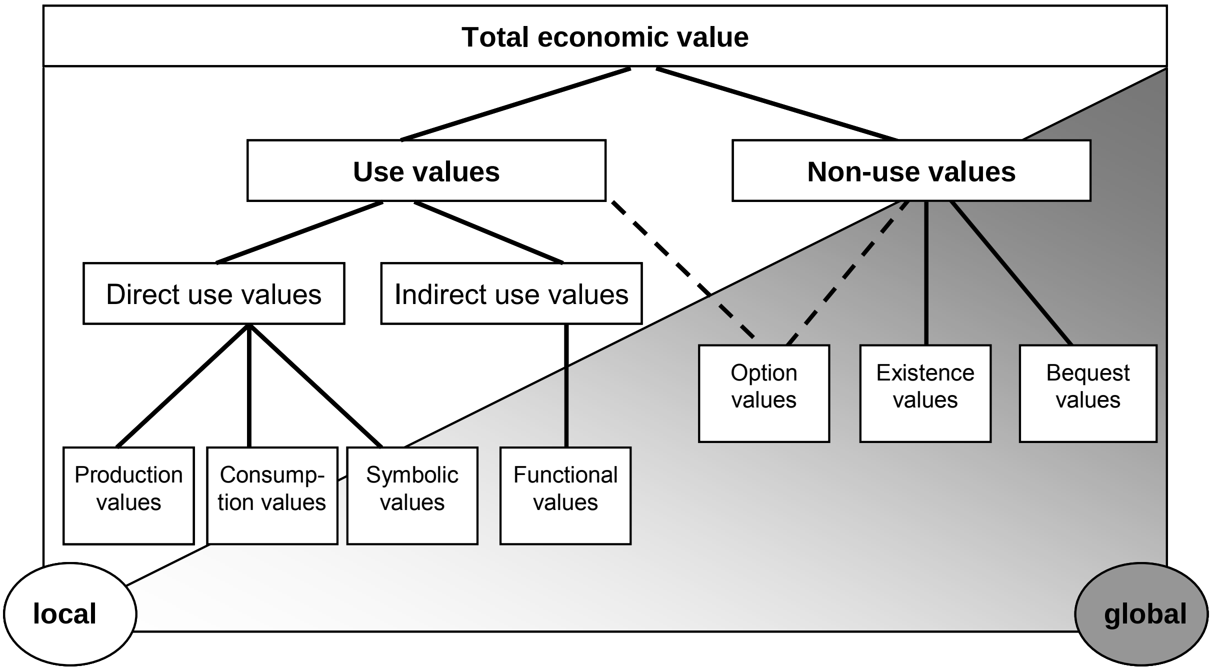
A closer look at the spatial distribution of the costs and benefits of biodiversity conservation enables a further assessment regarding the assignment of public goods and services to adequate governmental levels and potential needs for adjustment due to spatial externalities.

8.2 Local costs and global benefits of biodiversity conservation

8.2.1 The spatial distribution of conservation benefits

The total economic value of biodiversity results from adding up its partial values (see Figure 8.1). In economics, the various benefits of biodiversity conservation are usually looked at from an anthropocentric perspective, hence, the value of goods and services is gained on the basis of individual preferences (e.g. Pearce and Turner 1990). Market prices are the preferred indicator of economic value and are usually available for tangible goods and services. Positive and negative externalities, however, are rarely included in market prices. Reflecting the value of ecosystem services in market prices is also very difficult – if not impossible – in many fields of biodiversity conservation (Gowdy 1997; Millennium Ecosystem Assessment 2005b). There are various surrogate methods to reveal individual preferences for biodiversity goods and services when no market prices exist (Turner et al. 2003a; Turner et al. 2003b; Hein et al. 2006). They are widely used in economics, though one has to be aware of their implicit assumptions and shortcomings. Next to quantifiable economic values that at best allow an expression in monetary terms, safe minimum standards on the ecological side and cultural values obtained by deliberation on the social side are also part of valuing biodiversity.

The spatial distribution of conservation benefits has to be considered when decentralizing or centralizing competencies of biodiversity conservation (Döring 1998). The spatial distribution of beneficiaries usually increases when moving from direct to indirect use values and further on to option and existence values of biodiversity conservation (see Figure 8.1). At the most general level, conservation measures can have both local and global benefits (Perrings and Gadgil 2003). Even production values that can be captured at the local level may affect global benefits, depending on the scale and intensity of land uses and their impacts, such as local afforestation, which potentially leads to regionally changing weather conditions and helps to mitigate global climate change by carbon sequestration. There are also ecological feedback loops between local measures and higher level conservation objectives that require coordination at higher governmental levels.



Source: Adapted from Ring (2004a).

Figure 8.1: The spatial distribution of the benefits of biodiversity conservation.

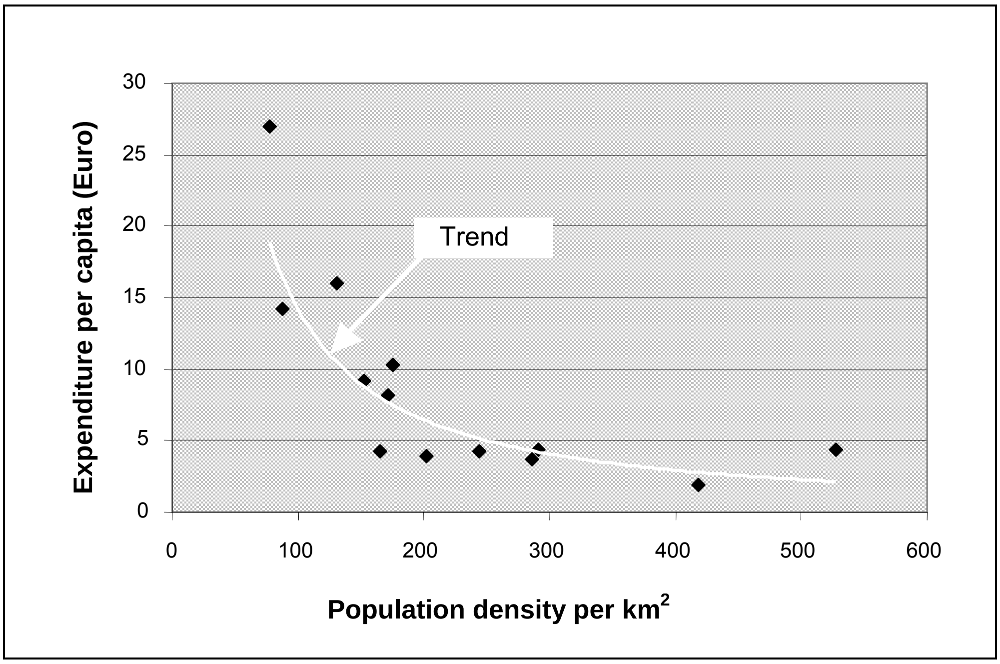
It is widely uncontested in economic and legal studies of federalism that especially the non-use values associated with natural resources justify the centralization of competencies (Revesz 2000; Oates 2001; List et al. 2002; Ring 2004a). Therefore, decisions on biodiversity conservation belong to the public tasks that have to be fulfilled at higher governmental levels, requiring centralized standard setting and policies. This is a fundamental difference compared to other fields of resource governance covered in this volume, particularly water and food. The need for central standards and principles of action in biodiversity conservation is reflected in the Convention on Biological Diversity and related activities. In the EU, it is carried out by several European directives that must be implemented by member states and must set a binding framework for biodiversity policies (Similä et al. 2006). On the one hand, the loss of biodiversity belongs to the very serious problems of global change. On the other hand, decentralized activities related to local land use have – if accumulated – a tremendous influence on the state of biodiversity worldwide. As a result, centralized standard setting still requires decentralized implementation of biodiversity policies.

Perrings and Gadgil (2003) address a number of reforms necessary to reconcile both local and global public benefits of biodiversity conservation. One of them is adjusting incentives to allow local communities to be rewarded and paid for their conservation efforts (Ring 2002).

8.2.2 The spatial distribution of conservation costs

From an economic perspective, biodiversity conservation is a public good. Its benefits accrue to overall society, that is, the wider political community, which ultimately relies on functioning ecosystems and their services provided to society. Therefore the state as the political representative of society has to set the framework conditions that guarantee the sufficient provision of the related public goods and services. Following the principle of fiscal equivalence, the costs of biodiversity conservation should also be borne by overall society. However, in reality the costs of biodiversity conservation are often distributed unequally among different groups of society.

First, we will consider public expenditures for nature conservation in relation to population density based on the example of Germany (see Figure 8.2). Empirical data for the various German states, or *Länder*, indicate that states with a lower population density bear the highest expenditures per capita (Ring 2004a). This is related to the fact that nature conservation normally takes place in space. The conservation value of landscapes often increases in remote and less inhabited areas where numerous large-scale reserves can be found.



Source: Bundesamt für Naturschutz, department II 1.1 (January 2004); data: Stratmann (2002).

Figure 8.2: Conservation expenditures of the German Länder (2001).

Second, the primary sector (agriculture, forestry, fisheries) is of extraordinary importance for the conservation and sustainable use of biodiversity. On the one hand, agricultural development in European landscapes has particularly contributed to the decline of biodiversity during the last century (Hampicke 1991). According to the results of the Millennium Ecosystem Assessment (2005a), the primary sector will continue to be responsible for the destruction and degradation of valuable habitats due to substantial land use changes on a global scale in the near and medium term future, as the result of further intensification of production mainly to meet increasing global food production needs. On the other hand, the costs of biodiversity conservation must largely be borne by the primary sector (Hampicke 2005). Current extensive and sustainable land use practices are rarely competitive in an economic sense. Whereas production schemes for intensification and enlargement of production in agriculture and aquaculture are often subsidized, payments for ecological services belong to a field in the process of development (Gutman 2003). Without broad societal support by rewarding ecological services provided by land users, the loss of biodiversity cannot be reduced or halted (Millennium Ecosystem Assessment 2005b; Beck et al. 2006).

Third, the unequal distribution of conservation costs also concerns the spatial distribution of nature reserves and related costs. These costs not only include measures for the conservation and sustainable use of biodiversity, but also the costs of long-term land use restrictions to be borne by communities. Depending on the protection category, local land use planning can be heavily restricted. As a result, most communities are not actively interested in the designation of protected areas. Taking Germany as an example once more, the lowest protection category of landscape reserves is still quite homogeneously distributed across the German states. However, there is an increasing concentration of protected areas for the categories of nature reserves, biosphere reserves, nature parks and national parks. Three out of 16 states (Mecklenburg-Western Pomerania, Lower Saxony and Schleswig-Holstein) hold 90 per cent of all German national park areas (Urfei 2002).

8.2.3 Adjusting local costs and global benefits for public and private actors

Biodiversity conservation is a public good with its benefits mainly accruing at centralized levels – be they national, European or global. The costs of biodiversity conservation are distributed both sectorally (primary sector) and spatially (conservation expenditure per capita and protected areas) in an unequal manner. This relationship requires a centralized standard setting. In this sense, the continuing and ambitious standard setting at the European level, for example, by the Habitats Directive and the establishment of the Natura 2000 network, is justified from the economic perspective of fiscal federalism. At the same time, the increasing trend to cofinance regional, state and national agri-environmental programmes as well as environmental measures in fisheries policies with European support can be underlined.

The arguments brought forward underpin the important role of the nationstate and the increasing role of international institutions for biodiversity governance. Substantial disadvantages may exist when federal states underestimate the essential role of the nationstate, for example, by assigning decision making powers predominantly to lower governmental levels. During the recent discussion on the reform of federalism in Germany, national conservation authorities and NGOs put forward strong arguments against considerations of potentially abolishing the federal nature conservation law (Hendrischke 2004; Ring 2004a). Nature conservation was and still is a rather weak policy field in the overall political arena and may – despite factual evidence – be sacrificed in the course of strategic political compromises. The political results of the reform follow this line of argument. Nature conservation will be part of the German Environmental Code that is presently in design. Compared to other environmental policy fields such as water resources and emission standards, however, the German states will have far-reaching legal entitlements to derogate from the national standards (SRU 2006).

Nevertheless, the extent to which the states will make use of these derogation rights in their state conservation laws remains to be seen.

Adjusting local costs and higher level benefits of biodiversity conservation can take many different forms (Millennium Ecosystem Assessment 2005b). The case study presented in the third part of this chapter is dedicated to protected area management, highlighting the necessity to integrate ecological indicators into intergovernmental fiscal relations. In Europe, it is common practice to compensate private land users for restrictions imposed by conservation policies. The same philosophy, however, applies to public actors on a local scale. Decentralized levels of government such as municipalities should be compensated for ecological services provided in the long run by having protected areas within their administrative boundaries. For the most part, this is still neglected in German and European fiscal relations (Ring 2002), rendering the following case study an innovative example to learn from.

8.3 The ecological ICMS in Brazil: Fiscal transfers for local ecological services

8.3.1 History and basic characteristics of the ICMS-E

Brazil is a federal country consisting of 27 states, each of which has an elected government with revenue-raising power. About 90 per cent of overall state tax revenues are based on the ICMS (Imposto sobre Circulação de Mercadorias e Serviços), a tax on goods and services similar to the value-added taxes in other countries (Loureiro and de Moura 1996; May et al. 2002). The ICMS also represents a relevant source of revenue for local governments because the Federal Constitution of Brazil decrees that 25 per cent of the revenues raised by this tax have to be allocated by the state to the local level of government. According to the Federal Constitution, 75 per cent of the total amount passed on to the municipalities is to be distributed in accordance with the share of the state ICMS that has been collected within that municipality. The state governments decide on the indicators for allocating the remaining 25 per cent.

During the early 1990s, the state of Paraná was the first to introduce ecological indicators alongside other indicators commonly used for redistribution by the states – such as population, geographical area or primary production (Grieg-Gran 2000). The new instrument of ICMS-Ecológico or ICMS-E (May et al. 2002) was created, through which allocation of revenues is based on environmental indicators.[3] Paraná started using ecological indicators in 1992, and soon other states followed this example. The states of Minas Gerais (1996), São Paulo (1996), Rondônia (1997), Mato Grosso do Sul (2002), Tocantins and Pernambuco (2003) started operating a similar system a few years later (Grieg-Gran 2000; May et al. 2002; Villar

[3] The ICMS-Ecológico is also known under the term 'ecological value-added tax'. However, this term is misleading from a public finance perspective. An ecological tax would be a tax assessed on the basis of ecological indicators. The ICMS-Ecológico uses ecological indicators for the allocation of its revenues. Therefore, economically speaking, the term 'ecological fiscal transfer' is more appropriate.

Martins 2003; CPRH 2003; Loureiro 2004). ICMS-E legislation also exists in the states of Amapá (1996), Rio Grande do Sul (1999), Mato Grosso (2001) and Rio de Janeiro (2007). Further Brazilian states are actively engaged in introducing ecological fiscal transfer legislation, including Santa Catarina, Espírito Santo, Goiás, Bahia and Ceará. In the states of Pará and Amazonas, the ICMS-E has been under serious discussion (Bernardes 1999; Freitas 1999; Loureiro 2001, 2005a; MMA 2002; Arantes 2006).

Each state is independent when making decisions about the indicators for distributing the 25 per cent of ICMS revenues to the local level. Therefore, different operating systems are in place regarding the consideration of ecological indicators, with one exception: all states with existing ICMS-E legislation have introduced the basic ecological indicator 'conservation unit' (CU), referring to the National System of Conservation Units in Brazil (Sistema Nacional de Unidades de Conservação). CUs include totally protected and restricted sustainable use areas that can be publicly managed (at the federal, state or municipal level), privately owned or managed by public–private partnerships. Eligible conservation areas need to be legally defined and registered in order to be considered (Grieg-Gran 2000).

The various categories of protected and sustainable use areas in Brazil – such as ecological research stations, biological reserves, parks, private natural heritage reserves (RPPNs) or environmental protection areas (APAs) – involve different degrees of protection and associated land use restrictions. Therefore, the states multiply the actual size of CUs with a weight reflecting its conservation value, with strictly protected areas at the top and sustainable use areas at the bottom of the scale ($0 < \text{conservation weight} \leq 1$). Each state is autonomous in applying its own relative weights.

The revenues allocated to the local level are based on an ecological index that represents the weighted CUs within municipal boundaries. Again, the states use slightly different calculation systems, but the general procedure can be described as follows (compare Grieg-Gran 2000; May et al. 2002): the municipal conservation factor is based on the total area set aside for protection in terms of CUs in relation to the total area of the municipality. The state conservation factor is given by the sum of all municipal conservation factors in the state. The ecological index is then calculated by dividing the municipal conservation factor by the state conservation factor. Finally, the municipal ecological index is multiplied by the total amount of ICMS-E revenues dedicated to CUs. For example, the ecological share of total ICMS revenues for conservation units is 2.5 per cent in Paraná, 5 per cent in Rondônia and Mato Grosso do Sul and 0.5 per cent in Minas Gerais and São Paulo (Grieg-Gran 2000).

Most states use a purely quantitative measure in terms of the quantitative amount of CUs. Although several of the states with ICMS-E legislation call for a quality index, in most cases they are only partially implemented if at all. The state of Paraná succeeded first in considering the quality of protected areas as part of their ecological index, rendering the implementation of the instrument more complex but also more effective in terms of conservation objectives.

8.3.2 Financial incentive to create new protected areas

The ICMS-E has been quite effective in encouraging the creation of new protected areas (Young 2005). In Paraná, for example, the total area in CUs grew by over 1.5 million hectares by the year 2000 (see Table 8.1), representing an overall increase of 165 per cent during the nine years since its introduction in 1992 (May et al. 2002). Municipalities in particular developed an interest in designating new public protected areas at the local level. The introduction of the quality evaluation of conservation units in Paraná also had a positive effect on the interest of municipalities to improve their management (Grieg-Gran 2000; Loureiro 2005b).

Table 8.1: ICMS-E in Paraná: The increase in public and private protected areas.

Protected areas	Until 1991 (ha)	Total by 2000 (ha)	Increase (%)
Public			
Federal	289,582	340,428	18
State	39,859	53,663	35
Municipal	1,429	4,169	192
Private/mixed			
APA	306,693	1,212,324	295
RPPN	0	26,124	
Other	0	53,607	
Total	637,563	1,690,315	165

Source: May et al. (2002) and own calculations. APAs can be designated at federal, state or municipal level. RPPNs can be designated at federal or state level.

In Brazil, protected areas can be designated by public institutions at federal, state and municipal levels, a procedure referred to as conservation federalism (Loureiro 2005a). They can be initiated by private actors or by a collaboration of private and public actors. As can be seen in Table 8.1, the most significant increase, both in absolute and relative numbers, is related to the category of APAs. These are sustainable use areas that can easily be created and involve relatively few land use restrictions. This is also reflected in the rather low conservation weight of 0.1 associated to this conservation category in Paraná, meaning that only 10 per cent of the actual size of an APA is relevant for ICMS-E transfers. In contrast to APAs, RPPNs are of higher conservation value (conservation weight 0.8).

Since the introduction of the ICMS-E, revenues can considerably change for municipalities with a large share of CUs. In Paraná, the total amount passed through to municipalities averaged over R$50 million (US$21 million) per year between 1994 and 2000 (May et al. 2002). There are positive examples like Piraquara (90 per cent of municipal area protected watershed, 10 per cent CUs) that increased their earnings by 84 per cent in 1995 (Loureiro cited in Echavarría 2000). Changing municipal revenues from the ICMS-E led to the creation of Brazil's first municipal consortium for biodiversity protection in 1995, which resulted in the foundation of the Ilha Grande National Park two years later (May et al. 2002).

8.3.3 Incentive for new forms of public–private interaction

The ICMS-E programme is clearly targeted at the local governmental level and its public actors, for ICMS-E revenues only accrue to the municipality and not to the owner of the land. The incentive effect thus primarily addresses local public authorities. There is, however, another incentive effect to encourage public–private partnerships in terms of more environmentally sound land uses. Some of the municipalities have started supporting land users in managing CUs, including provision of staff, equipment and vehicles for managing the areas (Bernardes 1999; May et al. 2002). These activities are mostly related to a specific category of reserves, namely the RPPNs, a private reserve category specified in the national system of conservation units. RPPNs are owned and administered by a wide range of institutions, such as NGOs, industry or private landowners. These reserves are established and run by their owners, who are fully responsible for their maintenance and management (Manigel 2004).

Both Paraná and Minas Gerais have actively been promoting the establishment of RPPNs as part of an integrated public–private partnership in buffer zones surrounding public protected areas, representing an important link between totally protected areas and intensively used landscapes for agricultural production (cf. O'Riordan 2008). RPPNs originally represented a reserve category to be designated and handled at the national level by the Federal Brazilian Institute for the Environment and Renewable Resources (Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis). The states of Paraná and Minas Gerais established state legislation in the 1990s, allowing a less bureaucratic way of designating and handling the private reserves at state level. This step, together with the financial incentive effect of the ICMS-E, contributed to a significant increase in the number and size of RPPNs. By the year 2000, some 26,124 hectares were designated in Paraná and 34,069 hectares in Minas Gerais (Bernardes 1999). Due to the very active role of state environmental agencies in promoting private reserves, Paraná now holds the largest number and area of RPPNs (183 reserves with 36,928 hectares) protecting the remaining Atlantic Forest, one of the global hotspots of biodiversity (Mesquita 2004).

The ecological fiscal transfers actually motivated local governments to assist landowners in measures to protect and maintain the environmental quality of their areas, and also helped them to prepare the necessary registration documentation (Bernardes 1999). Drainage improvements as well as road maintenance and access improvement belong to the infrastructure support municipalities provide for landowners. Although RPPNs are eligible for rural land tax exemption, May et al. (2002) found out that this is not much of an incentive to private owners. The hopes for a link with municipal services and the prospects of ecotourism revenues are higher.

With the rising significance of RPPNs, their owners soon founded associations of RPPN holders to better organize themselves, improve the management of the reserves, and speak with a common voice regarding their interests. State conservation authorities (public actors) and conservation NGOs (hybrid organizations) actively cooperate with these new organizations or with single potential future RPPN holders (private actors) in pursuit of creating new RPPNs (Coneglian 2005; Loureiro 2005b). They join forces in talking to mayors to make them aware of the benefits from ICMS-E and encourage them to share at least part of the additional municipal revenues to help landowners manage the CUs (Mesquita 2004; Jordan 2005).

Nevertheless, the public–private partnerships related to RPPNs have been subject to critique. Due to the size of property and the volume of resources involved, large farmers were prioritized for the most part, leading to complaints that public money was being used to benefit a few large landowners (May et al. 2002), even though small landowners would also be interested in RPPN creation. In the meantime, new ways have been explored that also take smaller properties for RPPN creation into consideration, calling for a reduction of transaction costs associated with their management.

At a more general level, the ICMS-E has substantially improved relations between protected areas and the surrounding inhabitants (Bernardes 1999). As with numerous other places in the world, protected areas were primarily seen as an obstacle to local development (e.g. in Germany: Bauer et al. 1996; Stoll-Kleemann 2001). With this new instrument, local actors started perceiving them as an opportunity to generate revenue. For some municipalities, it was the additional revenue that first created a basic awareness of the existence of protected areas within municipal boundaries. In Minas Gerais, for example, only federal and state protected areas were considered during the first year of the ICMS-E's operation. Locally protected areas existed but still needed formal registration to be eligible for the programme. Municipalities reacted quickly to the new incentive programme, registering locally protected areas for inclusion in the ICMS-E programme during the second year (Grieg-Gran 2000).

Since the new revenues, the attitudes of both public municipal actors and local citizens towards protected areas have begun to change. However, this is greatly dependent on the communication strategies of local authorities. May et al. (2002) investigated the public–private relationships in municipalities belonging to the Ilha Grande National Park in Paraná, where the municipalities are actively communicating the benefits of the ICMS-E to the local population. In this way, the whole community perceives the financial importance of the ICMS-E. In São Jorge do Patrocínio, for example, the ICMS-E represented 17 per cent of the overall municipal budget in 1998 and 71 per cent of the total ICMS transfers in the year 2000. The local authorities make clear for what purposes the additional revenues are used, such as well drilling to provide drinking water, cleaning and landscaping of the urban area, garbage collection, landfills, environmental education or enforcement of land use controls in parks and APAs (May et al. 2002). Local communities are thus made aware of the link between revenues, public investments and protected areas within municipal borders.

Federal environmental institutions – including the environmental ministry – support the spread of knowledge about the effects of the ICMS-E in Brazil. Conservation NGOs, such as

The Nature Conservancy of Brazil, actively work together with local networks of NGOs to promote its introduction in states where the ICMS-E does not exist yet (Veiga Neto 2005). A variety of actors join forces – both public and private, ecologically and socially motivated. Because the ICMS-E is distributed in lumpsum payments, the revenues can be spent for conservation purposes but can also help to alleviate social and health problems or improve the educational situation in the municipalities.

8.4 Conclusions

Biodiversity governance involves different governmental levels and a variety of actors, including public authorities, private landowners and representatives from hybrid organizations. Even in the case of a purely hierarchical instrument, introduced and managed by public authorities, the ICMS-E in Brazil has shown how numerous new interactions crossing the public–private divide can follow. The rising significance of the private reserves is a good example of the blurring boundary between public and private spheres of biodiversity conservation (see Sikor et al. 2008). The importance of the ICMS-E's creation of these links, however, lies in the fact that there are few schemes in Brazil that directly compensate land users for ecological services on their land. In this way, the ICMS-E became the major instrument for internalizing spillover benefits of biodiversity conservation by compensating municipalities for local costs of bearing conservation measures and land use restriction.

The situation is different in the EU. A variety of agri-environmental and conservation programmes exist to compensate land users for environmental services (cf. Penker 2008). Here, instruments for compensating municipalities, that is, the local public side, for their restrictions in land use planning and barriers to generating income to fulfil municipal public functions are commonly missing, although environmental advisory councils have long asked for the introduction of such instruments (SRU 1996). Ideally, both types of instruments exist, internalizing conservation spillover benefits to the local level for both public institutions and the private sector. But in reality, there are different political cultures calling for mutual education. At least in some cases, it would certainly be more cost effective in Brazil to avoid the detour via the municipal government in compensating private landowners for their conservation activities. In Europe, by turn, there is a need to consider decentralized levels of government by introducing ecological fiscal transfers to the local level (Köllner et al. 2002; Ring 2002; Perner and Thöne 2005). So far, only Portugal has set up a fiscal transfer scheme that explicitly rewards municipalities for having designated Natura 2000 sites and other protected areas within their territories (de Melo and Prates 2007).

There are also lessons at a more general theoretical level. For decades environmental economics and the theory of environmental policy predominantly focused on environmental pollution associated with negative externalities and respective policy instruments for internalization such as taxes, charges and fees. For the sufficient provision of environmental goods and services, internalizing positive externalities is just as important as internalizing negative externalities. Comparative efforts still have to be dedicated to the analysis of positive externalities and adequate policy instruments to address them.

Regarding biodiversity governance, an important task is to acknowledge and value the spillover benefits, especially of both public and private local conservation efforts. Conservation management is mostly management related to land use, where local communities play a role that is not to be underestimated. Even though some conservation related measures are paid by higher level institutions, local actors ultimately carry the costs, at least in terms of land use restrictions. Financial instruments to compensate the local costs of biodiversity conservation are a minimum requirement for the sufficient provision of biodiversity related goods and services. They can be provided by both public and private actors, although different types of instruments are needed for addressing each of them.

Agri-environmental payments constitute a common instrument for compensating private land users for their additional costs of conservation. All too often, however, agri-environmental payments in environmental economic literature and in the media continue to be summarized as subsidies (see also Sikor et al. 2008), frequently involving a negative connotation of market distortion. In fact, subsidies are defined in public finance literature as public transfers without equivalent market relevant services by the recipient. There is a substantial need for better differentiation in this respect, with true economic and societal recognition of the farmer's performance. If farmers provide ecological services to society in terms of biodiversity conservation, thereby producing positive externalities, they should be paid by society for these services. Otherwise, as economic reasoning tells us, there will be an underprovision of the respective goods and services, because markets are unable to cope with externalities. In economic terms, the financial instrument used for payments of environmental services should no longer be referred to as a subsidy.

Intergovernmental fiscal transfers in federal systems constitute a suitable instrument for internalizing spillover benefits between public jurisdictions at different governmental levels. So far, they have rarely been used for environmental purposes (Ring 2002). Environmental federalism primarily deals with competencies of environmental decision making. This involves the question of devolution: which environmental goods and services should be provided by public institutions and which by private actors (e.g. Anderson and Hill 1996)? And if this decision goes for public provision, the centralization/decentralization question arises (e.g. Harrison 1996): Which environmental standards should be set at which governmental levels? Analysing intergovernmental fiscal relations and respective policy instruments for their potential to compensate local governments for spillover benefits associated with the provision of environmental goods and services is still a large field to explore and develop. The Brazilian case study has shown the extent of local initiative and endeavour that still lies idle in states with no adequate compensation of local conservation services.

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9 Protected species in conflict with fisheries: The interplay between European and national regulation

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Abstract

Successful public conservation policies at various governmental levels have increased some populations of protected species to the extent that they are causing damage to human activities, like fisheries. As a reaction public authorities are developing biodiversity reconciliation policies. Finland and Germany have both created reconciliation policies for the conflicts between nature conservation and fisheries including a package of measures like management of population, support of technical measures and various types of compensation payments. All these measures are affected by European policy and law, though no special reconciliation policy has been adopted at European level. This article explores the options European regulation offers and the restrictions it imposes on Member States. Based on experiences with German and Finnish biodiversity reconciliation policies, the interrelationship between European and national regulation is elaborated, leading to suggestions for better coordination of reconciliation policies between different governmental levels.

[1] Jukka Similä and Riku Varjopuro are senior researchers at the Finnish Environment Institute, Helsinki, Finland. Randi Thum was senior researcher at the Department of Environmental and Planning Law, Irene Ring is deputy head at the Department of Economics, both at the UFZ Helmholtz Centre for Environmental Research in Leipzig, Germany.

9.1 Introduction

Public policy aiming at reviving the populations of threatened species has sometimes turned out to be so successful that the increased populations are causing damage to human activities such as agriculture, forestry and fisheries. Those significantly suffering from the damage have started to demand compensation for damage and control of protected species. Conservationists are afraid that the latter would have adverse effects on biodiversity. Thus there is need for reconciliation between the economic interests of using natural resources and the conservation of nature.

A reconciliation policy typically consists of different kinds of measures[2]. Three categories of measures to reduce the damage cover a major part of measures available: (1) management and control of protected species; (2) support for technical measures to prevent the damage; and (3) compensation payments. The use of each category of measures is linked to legal regulation in two ways: legal regulation at various governmental levels provides options for or restricts the use of measures and, often, the measures are linked to the adoption and implementation of new legal regulation.

In this article the relationship between different categories of measures and legal regulation is explored in the context of conflicts between increasing populations of fish-eating protected species and fisheries from a multi-level governance perspective. The cormorant (*Phalacrocorax carbo sinensis*), the otter (*Lutra lutra*) and the grey seal (*Halichoerus grypus*) can be mentioned as examples of species causing economic losses to human activities, namely in aquaculture and fisheries (Carss 2003; Glahn et al. 2000; Kranz 2000, p. 357ff.; Myšiak et al. 2004). First, we will explore what kind of options EC regulation offers and what kind of limitations it imposes on Member States aiming to reconcile the conflicts. Secondly, on the basis of experiences from Finland and Germany, we will discuss the different approaches used in these two countries. In Finland the major conflict relates to damage caused by the grey seal to coastal fisheries, whereas in Germany damages caused by the cormorant and the otter to aquaculture will be investigated. Thirdly, the interrelationship between European and national regulation will be explored and the potential for improvement discussed.

We have chosen the conflicts in Germany and Finland as examples, because they illustrate different kinds of conflicts of protected species with fisheries in terms of the nature of fisheries, species involved and ecological features of the area. The German conflict, which relates to inland fisheries, involves the cormorant and the otter, whereas the Finnish case relates to coastal fisheries and involves the grey seal. In addition, the countries represent both federalist (Germany) and centralists (Finland) legal systems. Furthermore, the cases show that implementation of EU policies may take very different forms and are interpreted in different ways in the Member States.

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[2] A wide range of measures is presented in Conover (2002).

9.2 Relevant European policy fields and laws

There is no European regulation specifically designed for the management of the conflicts under investigation here. Furthermore, certain policy fields potentially relevant, like land-use policies, are beyond the competences of European institutions. However, some European laws and regulations have links to measures for biodiversity conflict reconciliation taken at the national level. Certain parts of European regulation impose strict limits; others are flexible and provide opportunities for national policy-makers.

Apart from fisheries policy, which we will discuss in the next chapter, there are three relevant European fields for reconciliation of the conflicts covered, namely nature conservation law, European funds and State aid law. European nature conservation regulation is relevant for any domestic policy measures aiming to protect, manage or control the population of protected species. Certain European funds may be used to financially support national or regional policy measures, such as financial support for technical mitigation measures. European State aid law regulates the use of purely national subsidies, be it for technical measures or damage compensation payments.

9.2.1 The Common Fisheries Policy

Common Fisheries Policy (CFP) seems, at first glance, to be a very relevant policy field for the conflicts, because European institutions have a major role in designing fisheries policies in Europe, significant financial resources are devoted to fisheries policy and the environmental dimension has recently become stronger in CFP.

Despite this development, only few instruments of fisheries policy are directly relevant for conflicts arising from the increase of the populations of protected species. Here, the crucial question is: what are the impacts of increased populations of protected species on commercial fisheries or aquaculture and how can conflicts arising from these impacts be reconciled? Following recent developments of the CFP, the relationship between protected species and fishing is, instead, mainly seen from another, though very important perspective: what are the threats caused by fishing and fishing activities to the conservation of fish resources, or to the marine ecosystem?[3]

However, the CFP does have some relevance for our type of conflict. First, the CFP aims to regulate the overall fishing effort, but it has been documented that the CFP has largely failed in its conservation goals (e.g. Payne 2000; Symes 2005; Hentrich and Salomon 2006). Many of commercially important fish stocks have declined in the European sea area which, in turn, has immediate consequences in terms of the amount of fish available for natural predators, like seals. Fishermen and aquatic predators are competing for the same resources and many fisheries may, in fact, cause conflicts between protected species and fisheries as has

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[3] This point of departure is expressed in different recitals and Articles of Council Regulation (EC) No 2371/2002 of 20 December 2002 on the conservation and sustainable exploitation of fisheries resources under the Common Fisheries Policy, OJ 2002 L 358/59.

been indicated in many studies (Pauly et al. 2002; Gislason et al. 2000; Sinclair et al. 2002). In this way the CFP is generally relevant for the conflicts, but in none of the cases studied in this article fisheries poses a threat to these predators. Since cormorant and seal populations are growing fast, shortage of food is not a limiting factor. With regard to otter, pond farming is not a limitation but a precondition for the otter population by maintaining suitable habitats. Second, certain policy measures of the CFP are specifically relevant. For example, species like the grey seal involved in the conflicts may benefit from the by-catch regulation. The aim of the by-catch regulation has been two-fold from the inception of the CFP: to protect juveniles and to protect marine ecosystems by improving the inter-species selectivity of fishing gear (Long and Curran 2000, p. 18). With regard to the conflicts arising from the damage caused by protected species more selective fishing techniques may, in certain circumstances, have a double benefit: techniques may protect the protected species and reduce the damage at the same time. This is the case, for example, with regard to so-called seal-proof trap-nets used in the Northern Baltic Sea. However, the issue of selective techniques is not correspondingly relevant for aquaculture. Furthermore, even if the fishing gear could be developed to catch just the targeted species, the number of protected species, like seals, affects the size of fish populations and thus, the size of the fish catch. In other words, the conflicts addressed in this article would still persist after solving the by-catch problem.

In addition, CFP may contribute to the reconciliation policy by making financial resources available. Hence, the Financial Instrument for Fisheries Guidance (FIFG) is an interesting instrument for those making reconciliation policies, though the conflicts concerned obviously have not been in the minds of the drafters of this instrument. We will later discuss FIFG in the context of European funds.

9.2.2 Nature conservation regulation

The reconciliation of the conflicts arising from increased population of protected species may require the control of the population. However, such control has to fulfil the requirements laid down in the nature conservation law. A mitigation measure contravening nature conservation law is prohibited unless it is justified under the derogation rules. According to the Habitats Directive mitigation measures, which involve deliberate capture or killing, deliberate disturbance, deliberate destruction or taking of eggs from the wild, or deterioration or destruction of breeding sites or resting places are prohibited with regard to strictly protected species in Annex IV a) of the Habitats Directive.

A Member State may, however, allow mitigation measures, if the conditions as laid down in Article 16 of the Habitats Directive or Article 9 of the Birds Directive are met. With regard to the Habitats Directive, a mitigation measure must first fulfil two general conditions: there is no satisfactory alternative and mitigation is not detrimental to the maintenance of the population at a favourable conservation status. Secondly, it must aim to one of the legitimate purposes listed in the Article. Among them are the prevention of serious damage (e.g., to agriculture, forestry and fisheries) and the “interest of public health and safety or other imperative reasons of overriding public interest, including those of a social and economic nature”. Alternatively, it is possible to allow, under strictly supervised conditions, on a selective basis and to a limited extent, the taking or keeping of certain specimens of the strictly protected species in numbers specified by the competent national authorities. The Birds Directive has slightly different derogation rules. It stipulates that derogation, provided that there is no other satisfactory solution, may be allowed for certain legitimate purposes (like damage to fisheries) or to permit, under strictly supervised conditions, on a selective basis the capture, keeping or other judicious use in small numbers.

There are two relevant differences between the Directives. First, there is no reference in the Birds Directive to “other imperative reasons of overriding public interest, including those of a social and economic nature”. Secondly, the possibility to permit, under strictly supervised conditions, on a selective basis and in small numbers is limited to “the capture, keeping or other judicious use”. It is doubtful whether protective hunting could be considered as “judicious use”. If not, it would mean that the only derogation available for the reconciliation would be the prevention of serious damage to fisheries. Furthermore, on the basis of German experience, as will be discussed later, the seriousness of the damage is relative to the economic sector (fisheries), not to an individual farmer or fisherman. This interpretation would significantly limit the possibility to use this derogation, because often damage is serious “only” in relation to an individual operator. Apparently this is why the derogations rules of the Birds Directive are considered to be stricter in comparison to the Habitats Directive (Jans 2000, p. 421) and why the provisions in the Habitats Directive are described as “relatively loosely drafted” (Krämer 2000, p. 141f.).

Having said this and although European Court of Justice has required strict transposition of derogations and precise application of them (Jans 2000, p. 417), it must be noted that these provisions have not, in practice, prevented Member States from taking rather extensive measures if needed. For example, Denmark has recently started to take measures with regard to cormorants, which are rather extensive locally, but apparently do not yet have significant impacts at population level (Olesen 2005). To which extent these measures fulfil the requirements of the Directives is still untested in the Court. At this point one can only conclude that the Directives – and the Birds Directive even more so – set strict limits for population control measures.

9.2.3 State aid regulation

Financial aid for reconciliation measures may be considered to be State aid under European law. In practice, State aid rules have limited the possibilities of Member States to choose those reconciliation measures they have deemed appropriate to reconcile the conflict by forcing a Member State to adopt temporal measures, which can not do more than alleviate problems for a couple of years. Here the basic rules of State aid law are presented and the limitations arising from the rules are discussed later. The starting point is that any aid fulfilling all the following four criteria is considered to be State aid: (1) aid is granted from public funds, (2) it is selective (favours certain undertakings or the production of certain goods), (3) it distorts or threatens to distort competition and (4) it has trade effects between Member States

(Alkio and Wik 2004, p. 780ff.). From the guidelines on State aid in fisheries and aquaculture[4] it can be learned that State aid may take many forms. Any measure that entails a financial advantage in any form whatsoever funded directly or indirectly from the budgets of public authorities (on any administrative level) or from other State resources is considered to be State aid. Thus, this criterion is easily fulfilled. The third criterion is also easily fulfilled because the mere threat that competition is distorted is enough. Instead, the fulfilment of the second or fourth criterion is not so self-evident. Not all aid has trade effects between Member States. However, the most interesting criterion for a designer of reconciliation policy is the second one. If a compensation scheme is of a general character, which does not favour certain undertakings or the production of certain goods, it should not be considered as State aid assuming that there is no overcompensation. Overcompensation would fall under the third criterion.

The immediate consequence of considering a measure to be State aid is that the Commission must be notified. The Commission will then decide if the measure is compatible with EU rules. The Member State may not put the proposed measure into effect without the approval of the Commission. In many sectors minor State aid is exempted from the notification obligations provided that it fulfils certain requirements under the so-called *de minimis* rule. Recently the Commission has adopted a regulation on the application of *de minimis* rule in the agriculture and fisheries sector, which will be in force from January 2005 until December 2008. Aid not exceeding 3,000 Euro over a period of 3 years for any single enterprise and not exceeding the national ceiling (7,287,000 Euro for Germany and 460,200 for Finland) set for cumulative amount of aid granted is exempted from notification.[5]

Article 87(2) of the Treaty includes a list of aid that is always considered to be compatible with the common market, meaning it is always acceptable. Of the three categories included in this list only one seems to have a connection to reconciliation policy, namely aid to make good the damage caused by natural disasters or exceptional occurrences. As will be discussed later, using this as a basis for allowing State aid is problematic.

Article 87(3) of the Treaty includes a list of categories of aid, which the Commission may consider compatible with the common market. This list includes 5 different categories. Furthermore, the fisheries guidelines define more specifically what kind of State aid may be granted. However, it is hard to find any category of permissible State aid, which would be generally applicable to the conflicts in our interest. However, some of them could turn out to be relevant under certain circumstances.

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4 Guidelines for the examination of State aid to fisheries and aquaculture OJ 2004 C 229/5.

5 Commission Regulation (EC) No 1860/2004 of 6 October 2004 on the application of Article 87 and 88 of the EC Treaty to de minimis aid in the agriculture and fisheries sectors, OJ 2004 L 325/4, The general de minimis regime is established by Commission Regulation (EC) No 69/2001 of 12 January 2001, on the application of Articles 87 and 88 of the EC Treaty to de minimis aid, OJ 2001 L 10/30.

9.2.4 European funds

During the programme period 2000-2006 there are four EU Structural Funds: the European Regional Development Fund (ERDF), the European Social Fund (ESF), the European Agricultural Guidance and Guarantee Fund (EAGGF, Guidance Section) and the Financial Instrument for Fisheries Guidance (FIFG), to support the development of EU Member States. So far, there are no instruments specifically aiming to reconcile biodiversity conflicts. However, the financial instruments for agriculture (EAGGF, Guidance Section), fisheries (FIFG) as well as environmental and nature protecting measures as part of EAGGF, Guarantee Section, are of potential relevance and these can be used in designing conflict reconciliation as will be shown below.

The importance of EAGGF for conflicts between nature conservation and the use of biological resources lies in its potential to support agri-environmental measures including environmentally sound pond farming. Furthermore, support for "improving processing and marketing of agricultural products" (excluding products made of fish) is given.[6]

The Financial Instrument for Fisheries Guidance (FIFG) is the other and more widely applicable structural fund in the fisheries sector. Funded measures are required to[7] contribute to achieving a sustainable balance between fishery resources and their exploitation; strengthen the competitiveness in the sector; improve market supply and the value added to products or contribute to revitalising areas dependent on fisheries and aquaculture.[8] Areas for promoted actions are listed in Article 2(3)[9]. The following fields could be relevant for biodiversity conflicts: protection of marine resources in coastal waters, aquaculture, operations by members of the trade (for example operations to prevent fish stocks from damage), and innovative actions and technical assistance.

The single fields to be funded are described in more detail in Council Regulation No 2792/1999[10]. For example, Member States may take measures to encourage capital investment aimed at the protection and development of aquatic resources and aquaculture.

9.3 National regulation relevant for reconciliation

In this section we will explore how measures for conflict reconciliation have been incorporated into domestic law of two EU Member States, namely Germany and Finland. We will

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6 Art. 25–27 of Council Regulation (EC) No 1257/1999 of 17 May 1999 on support for rural development from the European Agricultural Guidance and Guarantee Fund (EAGGF) and amending and repealing certain Regulations, OJ L 1999 L 160/80.

7 Article 2 Council Regulation (EC) No 1263/1999 of 21 June 1999 on the Financial Instrument for Fisheries Guidance, OJ L 161/54.

8 Article 1 of the Council Regulation (EC) No 1263/1999, ibid.

9 Council Regulation (EC) No 1263/1999, ibid.

10 Council Regulation (EC) No 2792/1999 of 17 December 1999 laying down the detailed rules and arrangements regarding Community structural assistance in the fisheries sector OJ L 337/10.

cover all the three categories of policy measures mentioned: management and control of protected species, support for technical measures and compensation schemes. In addition, we will study how EU rules have modified and supported domestic legislation and what kind of implementation problems have arisen.

9.3.1 The case of Finland

The conflict in Finland concerns the conservation of the grey seal and coastal fishery. The fishery in question is a small-scale coastal fishing with gill-nets or trap-nets. It is typically a multi-species fishery targeting species like white-fish (*Coregonus lavaretus*), salmon (S*almo salar*) and pike perch (*Sander lucioperca*). Seals cause economic losses to fisheries by taking fish from nets and by breaking nets to the extent that it is considered a serious threat to coastal fishing (MoAF 2002). A growing grey seal population (Harding and Härkönen 1999; Helle et al. 2005) started to cause losses to coastal fisheries in the 1990s and the losses have gradually increased. When the problem became serious fishermen demanded that the State should start mitigating activities. A right for protective hunting of seals in order to minimise damage was introduced in 1997 and it was followed by economic compensation in 2003 and economic support for technical measures in 2004. These mitigation measures and policy instruments are described below in more detail.

9.3.1.1 Protective hunting of species to minimise damage

Protective hunting of seals as a mitigation measure has been used in Finland since 1997, when it was reintroduced after 15 years of total ban. At first hunting was only meant to target individual seals known to take fish from fishing gear. In the first year licences to shoot 30 grey seals were granted. Gradually the number has increased, so that during the hunting year 2005/2006 it is 635 individuals (MoAF 2005). After the first years, protective hunting has been conducted essentially as “hunting” of seals (i.e. carcasses are utilised), not just as killing seals to minimise damages.

The hunting is controlled by the Hunting Act with a quota system that determines regionally how many seals can be shot during a hunting year. The Ministry of Agriculture and Forestry (hereinafter ‘MoAF’) determines the size of regional quotas based on the scientific advice provided by the Finnish Game and Fisheries Research Institute. Regional Game Management Districts grant hunting licences and monitor hunting. The hunters have an obligation to report each kill and the hunting is stopped when the quota is full. In addition to the quota, hunting is regulated by a closed season to protect breeding and by the technical requirements of hunting. This measure meets the requirements of derogation rules under the Habitats Directive.

9.3.1.2 Compensation schemes

There are two compensation schemes in Finland to cover damage that seals cause to professional fishermen[11]. These are a compensation for loss of catch and an insurance that covers damage to nets.

A compensation scheme for loss of catch due to seals for professional fishermen was first planned as a permanent compensation scheme to be funded entirely from national sources, but the European Commission did not accept it due to EU State aid law. Hence, Finland modified the compensation scheme to cover only damage sustained in the years 2001 and 2002. The Commission stressed in its decision[12] that the compensation scheme was allowed due to its temporary nature and due to the fact that the damage was considered exceptional (unexpected increase in seal population). The Commission noted that according to legal practice[13] compensation paid for damage caused by public bodies to private persons is not State aid. However, it claimed that in this case the nature of the compensation was different. Taking also into consideration that it had trade effects, it was to be considered State aid.

The amount of compensation was calculated as a function of average loss of catch in a given fishing ground, days of fishing, type and amount of nets, average catch per unit and a price of the fish species in question. However, fishermen had to meet the first 250 Euros per year themselves. Compensation totalling 7.4 million Euros was paid to professional fishermen who had suffered over 20 % loss of catch.

Since 1930s there has been an insurance system for any physical damage to fishing gear of professional fishermen, including seal-induced damage, which is, in fact, the most common physical damage today in many coastal areas (e.g. Österbottens fiskeriförsäkringsförening 2003). The insurance system is partly financed by insurance premiums and partly by the state.

Compensation is granted by regional insurance associations. Fishermen have to cover part of the damage themselves. It is 25 % of the estimated damage in the case of gill-nets. Regarding other fishing gear the deductible proportion varies from 5 % to 25 %, depending on the value of the damage. State subsidy to the insurance association is 40 % of the damage on gill-nets and in case of other gear it is 40 % of the amount paid up to 504 Euros and 90 % of the rest of the compensation (Österbottens fiskeriförsäkringsförening 2003).

The insurance mechanism is at the moment under review from the perspective of State aid law. The process was initiated due to the change of the relevant State aid guidelines in 2004, which resulted in a general overview of all possible State aid mechanisms. According to pre-

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[11] Professional fishermen are those who get 30 % or more of their income from fishery.

[12] N102 /2001 Finland, 7.5.2002, C(2002) 1598 final.

[13] Commission referred to ECJ, Judgement in case C-106/87 – Asteris AE and others v Hellenic Republic and European Economic Community [1988] ECR I-5515.

liminary and indicative information the Commission is inclined to think that the insurance mechanism violates State aid law[14].

9.3.1.3 Support for technical measures

Technical measures to reduce damage to catch and fishing gear have been developed in Sweden and Finland and the most promising of these are modified trap-nets (e.g. Suuronen et al. 2004). In Finland, the development and testing of fishing gear has taken place mainly in projects of a fisheries research institute funded from national or Nordic sources. In addition, regional fishermen's associations have carried out projects to test the trap-nets and to inform fishermen about the new technology. These projects have been funded nationally, but in some cases co-financed by the EU through a programme called Community initiative concerning trans-European cooperation intended to encourage harmonious and balanced development of the European territory (INTERREG).

In 2004, the MoAF announced a change of policy involving the use of European funds for technical development. The new subsidy helps fishermen to invest in "seal-proof" trap-nets, which are expensive, but technically effective. FIFG funds cannot be used to subsidise investment in fishing gear and therefore permission from the Commission was requested before adopting the new regulation. Permission was given because this gear can be used in selective salmon fishing and thus its use enhances protection of naturally breeding salmon. In other words, protection of salmon and protection of coastal fishing from seals are linked in this decision (MoAF 2004, 2005)

The subsidy became available in 2004 when professional fishermen were allowed to apply for it from the regional fisheries authorities. Altogether 90 fishermen submitted applications for subsidies to purchasing 250 trap-nets totalling in an amount of 2.5 million Euros. The MoAF gives the applications final approval and the money is paid to fishermen after they have purchased the trap-net. They will be subsidised only once and the subsidy is 70 % of value of the first two sets of gear and 50 % for the rest (MoAF 2005).

9.3.2 The case of Germany

In Germany, the focus is on cormorants and otters, two fish-eating protected species in conflict with aquaculture. The region investigated is situated in Upper Lusatia in the German state of Saxony. Its landscape is characterised by numerous artificial ponds for carp farming, allowing for a viable otter population, and attracting an increasing number of migrating cormorants (Seiche 2002). Otter damage to aquaculture is less relevant compared to damage caused by cormorants and herons (Sächsische Landesanstalt für Landwirtschaft 2003). However, this may become locally significant, especially when otters prey on carp in small ponds used for wintering. People sustaining damage are professional fisherman, people with fishery

[14] Information is based on personal communication with the Ministry of the Agriculture and Forestry (May 2005).

as a secondary occupation and amateurs. Other relevant stakeholders are recreational anglers, but they are not yet covered by policy measures addressing the conflict.

The German legal system is characterised by its federal structure. For the relevant policy fields of nature conservation and hunting, there is one federal framework law, which frames the different 16 more detailed and implementing State laws. Each State (*Länder*) further issues its own regulation and administrative ordinances regarding biodiversity conflict mitigation.

9.3.2.1 Exceptions from the protection status

Whereas in Finland the main legal provision for the conflict refers to hunting law, in Germany protection of species is predominantly provided for by nature conservation law (as part of the Federal Nature Conservation Act) and only to a lesser extent by hunting law. The binding federal regulation concerning species protection is very detailed and thus the laws of the various States (*Länder*) are largely uniform in this respect. In German conservation law, there are basically three categories of protected species: "strictly" and "specially" protected species, whereas the "strictly" status is the higher one (e.g., it is prohibited to disturb them in certain sites). The third category relates to "European birds". They are in general "specially" protected species, but partially their protection status is analogous to strictly protected species. Following this system, the cormorant clearly belongs among the European birds, whereas the otter is a strictly protected species.

The German legislator aimed to transpose the European derogation rules without imposing further restrictions of national origin. But he did not transpose all of the European derogations. The general derogation clauses, i.e. "the taking or keeping of certain specimens of strictly protected species under strictly supervised conditions, on selective basis and to limited extent" (Habitats Directive) and "to permit under strictly supervised conditions, on a selective basis, the capture, keeping and other judicious use of certain birds in small numbers" (Birds Directive) are not transposed into German law. Hence, German law may be considered to be stricter than European law.

The exception to prevent damage to the fisheries sector may be granted either in every single case or, provided that the animals concerned do not belong to strictly protected species, generally by a State law. This means that exceptions for the strictly protected otter may only be granted by a single permission, whereas exceptions for the cormorant may also be granted by statutory ordinances ("cormorant regulation"). Regarding cormorants there is a marked difference between administrative practice and the judgements of the courts of justice. Authorities usually grant requested single permissions for shooting cormorants. For example, in the administrative region of Dresden, which is part of the German State of Saxony between 700 and 800 single permits were granted annually and some 600–700 animals were culled as a result of these permits.[15] Sometimes authorities refused to grant single permits. Appellants

[15] Data source: Saxon State Ministry for Environment and Agriculture (2004).

could then appeal to the courts against the refusal of single permits by authorities. In these cases, all courts appealed to refused requests for single permissions to cull cormorants, thereby confirming the authorities' decision, because the protective hunting could not prevent damage to the fisheries sector as a whole. Courts justified their decisions with the arguments that the connection between the presence of cormorants and damage was not yet clear and it was expected, that other cormorants would take over the free habitats. Moreover, in contrast to court judgements, several State authorities have allowed the protective hunting of cormorants by statutory ordinances ("cormorant regulations"). As a result of such a cormorant regulation 858 cormorants were culled in Baden-Württemberg in winter 2002/2003 (Thum 2004). In the case of cormorant regulations, the courts do not come into play, because there is no right to take the matter to court. This shows that in practice the strong demands for permits are not barriers for even far-reaching measures based on single or more general permissions. This also shows that there is no consistent implementation of legal requirements effected at the European level. Administrative courts and authorities use the same legal requirements in different ways and there is no possibility for harmonisation. For example the questions which damage could justify exceptions to the protection status of animals and how such damage can be correctly quantified (Seiche and Wünsche 1996) are still open.

In contrast to the protective seal hunting in Finland, the culling of cormorants is not perceived as hunting but as an exception to the protection status in order to minimise damage. Actually, for most protected animals population control by hunting is not allowed under German law. As a basic principle hunting law and nature protection law exist independently in parallel. Hunting is permitted if a species is subject to hunting law (defined in the Federal Hunting Act and the particular State Hunting Act) and if a hunting season is defined (Federal and State Hunting Season Regulation). Thus in principle every State (*Land*) is competent to determine which species may be hunted individually. However, both federal and State legislators are bound to European law. As long as the use of species is not permitted by Article 14 of the Habitats Directive or Article 7 of the Birds Directive, hunting of otters and cormorants is neither allowed in Europe nor Germany.

9.3.2.2 Support for environmentally sound agriculture

The Saxon Support Programme for Environmentally Sound Agriculture[16] is based on the European Agricultural Guidance and Guarantee Fund (EAGGF), section Guarantee. To use this fund, the States (*Länder*) set up development plans. The Saxon Development Plan for Agrarian Regions[17] includes the funding of agri-environmental measures, which are now part of the Saxon Support Programme for Environmentally Sound Agriculture. Its main importance in the context of conflicts concerning protected species lies in the funding for the main-

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[16] Richtlinie des Sächsischen Staatsministeriums für Umwelt und Landwirtschaft zur Förderung einer umweltgerechten Landwirtschaft im Freistaat Sachsen (UL) of 8 November 2000; RL-Nr.: 73/2000; last modified on 10 Juli 2003.

[17] Entwicklungsplan für den ländlichen Raum in Sachsen 2000-2006.

tenance of important habitats. The programme part "Nature Conservation and Protection of the Cultural Landscape" (NAK) supports the conservation of threatened, historically valuable ponds. All benefits are only granted within the scope of funds available and no legal claim can be made. In this context, Saxon aquaculture has been supported with 2.3 to 2.6 million Euro per year between 2000 and 2003, covering more than 95 % of Saxon pond area used for farming. NAK funds represent the major part of public support to aquaculture that adds up to approximately 20–30 % of the total gross income per hectare (Klemm 2001). One measure of the programme is unofficially called "otter bonus" and pond farmers are paid 103 Euro/ha per year for extra stocking of ponds with fish to provide a feeding habitat for the otter. Between 2000 and 2003, this single measure provided pond businesses applying it with overall 230,000 to 311,000 Euro per year in Upper Lusatia.[18] The otter bonus is the most important policy instrument in Saxony to reconcile the conflict between otter conservation and aquaculture. Mostly due to this instrument, the majority of Saxon pond farmers do not perceive the otter as a conflicting species any more, "the otter belongs to the landscape" (Zwirner and Wittmer 2004, p. 132). From an economic perspective, the NAK programme rewards aquaculture for the additional costs it carries due to environmentally sound production that in this way is born by society as a whole.

9.3.2.3 Compensation scheme for damage caused by protected species

Saxony is the only German State that pays for damage caused by wild animals. Contrary to the previously mentioned agri-environmental compensation scheme, the damage compensation scheme is only funded from the budget of the State of Saxony without any link to European funds. Both the Saxon environmental ministry and the State conservation authority do not see a link between this conservation-oriented damage compensation scheme and European State aid regulation, therefore no permission was requested from the European Commission when inaugurating the scheme. The basis for damage compensation is the "Compensation for Cases of Hardship Regulation". Payments require significant hardship for the owner or user of resources: damage exceeding approximately 100 Euro per hectare per annum (in agriculture and fisheries) or 50 Euro (in forestry) and the aggregated damages exceed 1,000 Euro (in agriculture and fisheries) or 50 Euro (in forestry). The amount of compensation is usually 60 % and may rise up to 80 % of the overall damage. Based on this regulation, Saxon damage compensation for cormorants ranged between 670,000 and 800,000 Euro in the years 2000 to 2002. In this way, it is the main instrument to compensate for cormorant related damages. However, pond farmers and respective associations still perceive the cormorant as the main conflicting animal with aquaculture, and measures for conflict mitigation involving population control measures are asked for by this stakeholder group (Zwirner and Wittmer 2004). For the otter, damage compensation payments in Upper Lusatia (highest otter densities in Saxony) ranged between 23,000 and 70,000 Euro between 2000 and 2002.[19] Therefore, the

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[18] Data source: Sächsische Landesanstalt für Landwirtschaft, Referat Fischerei (2004).

[19] Data source: Sächsische Landesanstalt für Landwirtschaft (2004).

previously mentioned otter bonus is the more important instrument regarding the otter. The damage compensation scheme still seems to be necessary for locally high damages. Pond farmers not sustaining great damage from otters could obtain restitution in kind from some district authorities. The basis for this procedure was a special guideline no longer in force. At the moment local authorities are looking for a new solution to cover small-scale damage.[20] Sociological analysis has shown that the existence of the various compensation instruments in Saxony is the most important reason why the conflict about the fish-eating vertebrates with aquaculture is calmed down (Zwirner and Wittmer 2004).

9.3.2.4 Financial support for technical measures to avoid damage

In Saxony, financial support is provided for measures aimed at avoiding damage caused by protected species in the framework of the Financial Instruments for Fisheries Guidance (FIFG) of the European Union. To use FIFG, the operational programme for Germany's Objective 1 region[21] was set up. Regarding aquaculture, the programme stresses the increasing number of cormorants. In Saxony the operational programme was implemented by the Support Programme for Aquaculture, which funds technical measures to protect fish stocks from cormorants, herons and otters. Support is given as direct project support for 60 % of the total costs. Measures currently supported are fences against otters, surging measures with nets against cormorants and anti-cormorant equipment. Between 2001 and 2003, only 7 projects have been supported in Saxony (from 3,000 to 78,000 Euro). The reason may be that the type of equipment utilised is very expensive and the remaining investment costs for the pond farmers are mostly too high compared to the return of an average pond farm (Klemm 2001).

9.4 Interplay between European regulation and national reconciliation policies

In this final section we aim to identify key problems obstructing effective reconciliation policies, related to the design of regulation as well as its implementation. The main focus is on the European level, for European regulation constitutes the binding framework to be considered and implemented by Member States. However, the national experiences are used as examples to illustrate the practical consequences of European regulation. We will discuss how the existing European regulation could be further developed and improved in order to better meet the challenges of biodiversity conflict reconciliation policy.

9.4.1 Nature conservation regulation

European nature conservation law covers most species potentially involved in conflicts. Many of those species are strictly protected, meaning that reduction of damage caused by protected species through methods of killing and disturbance of them are prohibited, though they may

[20] Sächsische Landesanstalt für Landwirtschaft (2004), personal communication.

[21] Operationelles Programm FIAF 2000-2006.

be allowed for reconciliation purposes under the derogation rules of the European nature conservation law.

In practice, Member States have adopted culling schemes, as in the cases of seals and cormorants, to prevent local damage. The culling schemes adopted in both countries studied have been somewhat restrained. Regarding cormorants, France pursues a very extensive culling policy, shooting tens of thousands of cormorants each year (Marion 2003), and Denmark has recently made its first attempts to regulate the species at the population level by way of egg-oiling in specially selected breeding colonies (Olesen 2005).

However, the legality of culling as a reconciliation measure has never been tested in the European Court of Justice, but some derogation rules of European origin have been interpreted by the courts of Member States, like those of Germany. Interestingly, the German courts have never granted permits to cull cormorants based on a strict interpretation of nature conservation law, whereas implementing authorities grant permits more easily within their realm of legitimate decision-making (Thum 2004). So far the German practice has formally been based on national law. However, given reasons and taking into account the reservations explained above, the German legal practice can be used at least as a rough indication of how courts could apply European law.

There are two important points to be raised with regard to different mitigation practices pertaining to all species. The first concerns the concept of “considerable damage” in German legal practice. This concept is considered to be relative to the economic sector (like fisheries), not to an individual farmer or farm. Because the individual claimants could not prove that the damage was considerable at the level of the whole economic sector, the authorities’ refusal to grant a permit to kill cormorants was confirmed by the courts. Secondly, the derogation measures, including protective hunting, can be allowed only if they can be assumed to be effective in relation to the purpose for which they are used. Because the effectiveness of culling or hunting species as a measure to prevent damage is often doubtful this requirement may turn out to be important. For example in the case of Finland, the hunting of seals aims to expel the seal from fishing grounds, not to affect the size of the population. However, it is not clear if hunting will have such an effect (see Westerberg et al. 2000). If further scientific evidence supports these doubts, the justification for using protective hunting as a means to prevent considerable damage to fisheries disappears. The same principle applies to cormorant culling in Germany, where it has influenced the court decisions.

The approach of the Birds Directive is stricter than that of the Habitats Directive. Both provide for a general mechanism to allow, in limited numbers and on a selective basis, derogations from general protection without restrictions explained above. However, the Birds Directive restricts this possibility only to “capture, keeping and other judicious use of certain birds”, whereas the Habitats Directive also allows “taking”. Because it is doubtful if killing a bird is “judicious use” of it, this derogation may not be available for culling schemes.

At political level, a sound reaction to the situation would be to consider if there is a need to change the conservation status of now abundant species. If the derogation rules do not allow reasonable reconciliation policy, the problem is not necessarily the courts intepreting the rules precisely and specifically, but the wrong conservation status. In fact, the increasing number of cormorants already caused an amendment to European law in 1997: Member States are no longer required to protect the habitats of cormorants by establishing special protected areas. Due to the stable population level of cormorants, one could envisage a further change in their conservation status and allow controlled hunting by removing this species to Annex II of the Birds Directive. Similarly, if the present steady growth in the seal population (Helle et al. 2005) continues, the issue may become topical also in the case of Baltic Sea grey seals.

9.4.2 State aid regulation

State aid law has affected reconciliation policies in Member States. In the countries studied there are three different policy measures aiming to grant compensation from national sources for damage caused by protected species: (1) the compensation of loss of catch in Finland, (2) the Finnish insurance system that compensates physical damage to fishing gear, and (3) the Saxon hardship compensation regulation. As a result of negotiation between the Commission and Finland the first scheme was made temporary, covering only two years. The other Finnish instrument is currently under review and on the basis of preliminary information available, the Commission tends to think that the scheme violates State aid law. The permanent German regulation on compensation for cases of hardship is in place and has never been notified as a State aid to the Commission. The Saxon State Ministry for Environment and Agriculture does not consider these payments as subsidies subject to European State aid law.[22] In addition to the state of Saxony, some other countries compensate damage to fisheries caused by wild animals. For instance, Sweden compensates coastal fishermen for damage caused by grey seals (Bruckmeier and Höj Larsen 2005) and a compensation scheme for cormorant damage is in place in the Province of Ferrara in Italy (Moretti et al. 2005). Apparently only the Finnish schemes have been notified according to the State aid rules. Furthermore, in Finland, too, there are compensation schemes for damage caused by wild animals in other economic sectors than fishery. Hence, not all the schemes have been investigated by the Commission – not to mention the European Court of Justice.

Until now one compensation scheme, namely the compensation for loss of catch in Finland, has been considered to be State aid and in that case the aid was allowed, though only temporarily. In addition the Finnish insurance system is under review. One may ask what the legally relevant difference is between the Finnish schemes considered to be State aid and others. In this regard we draw attention to criterion 2 (selectiveness) explained above. A compensation scheme, which does not favour certain undertakings, could be permanent.

Even if aid is State aid, it can be compatible with the common market, and thus, accepted. Certain aid is always compatible with the common market and the Commission may consider others to be compatible. For example, aid aiming to make good the damage caused by natural

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[22] Saxon State Ministry for Environment and Agriculture (2005), personal communication.

disaster or exceptional occurrences is always compatible and the Finnish scheme was temporarily allowed on this ground. Furthermore, the Commission may consider compatible other aids belonging to one of five categories listed in the Article 87(3) of the EU Treaty. However, none of these categories is visibly related to reconciliation policy, though some of them may in certain circumstances turn out to be relevant.

Any aid deemed to be State aid must be temporary to be compatible with the common market. However, for the purposes of reconciliation policy this may be problematic. The economic damage caused especially by seals and cormorants is nowadays greater than ever and there is no mitigation measure available to bring a solution to this in the near future. Extensive hunting, which has previously reduced the population of protected species to an unacceptably low level, is – and should remain – out of the question due to nature conservation goals. Fishermen and protected species will compete for the use of same resources in the future, too. Thus, the problem is likely to persist.

In practice, there may be only few options for mitigation measures to reduce the economic losses caused by protected species. This is especially the case for very strictly protected and seriously endangered species such as the otter. State aid regulations should generally support nature conservation and species protection that are ecological services to society as a whole. In our view State aid law should not prevent Member States from creating national compensation schemes related to damage caused by protected species when this option is supported by serious reasons. However, market distortions and trade effects should be avoided as far as possible and the preference should always be for preventive measures that may result in a permanent mitigation of damage and thus conflicts. However, it is unlikely that trade effects would be on a large scale and, therefore, reconciliation of the conflicts should take preference over trade effects. Thus, there are grounds to suggest that this issue should be discussed when the relevant State aid guidelines are next reviewed.

9.4.3 European funds

The various European funds can be used for the reconciliation of the conflicts. For example, FIFG funding has recently been directed at the investments in seal-proof fishing gear in Finland. The Saxon “otter bonus”, which is a part of an EU-based agri-environmental programme, is another example. As part of this programme both the maintaining of valuable habitats, i.e. fish-ponds of high environmental quality, and the provision of feeding habitats for otters are seen as ecological services to society. Thereby they are justified on economic grounds (Hofman et al. 1995).

However, a lack of special provision in the fund regulations regarding reconciliation policies and measures may lead to problems. Without such a provision it may become complicated to design proper measures, which really can achieve reconciliation and, at the same time, meet all the other requirements of the regulation. For example, many instruments under European funds are geographically limited and these restricted areas are not necessarily the same as those of the conflicts. Thus, a clarification of the regulation concerned in this point would be preferable. This becomes even more important because the laws on European funds and State aid are, from a policy-making point of view, interlinked. The State aid law does not apply to European funding. Thus, a proper European funding regulation may help to overcome possible problems related to State aid law. Because State aid law does restrict the possibilities of the Member State to adopt national compensation schemes, an alternative policy response to the change of State aid law discussed above would be to create a European funding mechanism for the compensation of damage. Furthermore, European Structural Funds are especially well suited for funding activities to develop or invest in preventive measures.

9.4.4 Need for better coordination

Effective reconciliation of conflicts arising from damage caused by protected species requires several measures often administered by different administrative sectors and regulated by different laws. At the moment reconciliation policy is to a large extent a national matter in Europe, though certain fields of European law and policy are relevant. They both provide options and set limitations for reconciliation policies as discussed above. Hence, there is a need for coordination at both levels.

The need for coordination is obvious in both cases studied. In Finland, the lack of coordination of policy measures has impeded effective management of the conflict in spite of 10 years of mitigation. In Germany the federal structure stresses the importance of coordination for both nature conservation law and funding mechanisms differ from one State to another. However, this may be both an advantage and a disadvantage, depending on the specific conflict. On the one hand, the federal system in Germany allows for spatial differentiation and regionally adapted management of conflicts, providing options for increased cost-effectiveness. On the other hand, transboundary problems require coordinated action, as is the case with migrating or spatially more widely distributed species (Ring 2004).

European institutions could have a more active role in coordination. First, many conflicts have implications for Europe as a whole, as is obvious in the cases of migrating species like cormorants or species moving in large sea areas crossing borders of many Member States like the grey seal. European institutional structures and funding mechanisms could provide a sound basis for coordination of activities in the Member States. Second, European law, like nature conservation law and State aid law, is anyhow involved in national reconciliation policy and possible conflicts between law and the needs for reconciliation could be avoided by a more active Europe-wide policy. Furthermore, through coordination it would be possible to avoid or minimise different interpretations of European law at Member State level. Different interpretations may even occur within a Member State, as the case of Germany shows.

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