Incentive Effects in Ecological Fiscal Transfers

Evidence based foundations for policy advice

DISSERTATION

zur Erlangung des Grades

Doktor der Wirtschaftswissenschaft (Dr. rer. pol.)

der Juristischen und Wirtschaftswissenschaftlichen Fakultät
der Martin-Luther-Universität Halle-Wittenberg

vorgelegt von
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Zusammenfassung

der Dissertation

Incentive Effects in Ecological Fiscal Transfers

von Nils Droste

zur Erlangung des Grades Doktor der Wirtschaftswissenschaft (Dr. rer. pol.)
der Juristischen und Wirtschaftswissenschaftlichen Fakultät
der Martin-Luther-Universität Halle-Wittenberg

Diese kumulative Disseration untersucht mittels empirischer Analysen die Anreizwirkungen ökologischer Finanzausgleichsysteme auf die Ausweisung von Schutzgebieten. Die Arbeit zeigt damit erstmals systematisch die Auswirkungen von ökologischen Finanzausgleichsystemen auf und generiert Grundlagen für eine evidenzbasierte Politikberatung. Unter Betrachtung spezifischer institutioneller Rahmenbedingungen werden Handlungsempfehlungen für die Ausgestaltung entsprechender Mechanismen entwickelt und ihre Funktion in der Umwelt- und Naturschutzpolitik beleuchtet. Es werden konkrete Vorschläge für (inter-)nationale Politikgestaltung ökologischer Finanzzuweisungen entwickelt.

Für die theoretische Fundierung der empirischen Arbeiten entwickelt die Dissertation ein mikroökonomisches, finanzwissenschaftliches Modell, wie ökologische Zuweisungen im Finanzausgleich relative Kosten der Erbringung von Naturschutz ändern und damit finanzielle Anreize zu einer erhöhten Bereitstellung setzen können. Somit werden überprüfbare Hypothesen für die folgenden empirischen Arbeiten entwickelt.

Mittels einer mikroökonometrischen Paneldatenanalyse wird daraufhin am Beispiel *Brasiliens* gezeigt, dass Bundestaaten mit ökologischem Finanzausgleich durchschnittlich signifikant mehr Gemeindeschutzgebiete in Prozent der Flächen ausgewiesen haben. Damit wird grundsätzlich deutlich, dass Finanzuweisungen für Schutzgebietsflächen Gemeinden zu mehr Naturschutz bewegen können. Mittels einer Bayesianischen Zeitreihenanalyse wird am Beispiel *Portugals* gezeigt, dass nach Einführung des ökologischen Finanzausgleichs das Verhältnis von Gemeindeschutzgebieten zu nationalen Schutzgebieten signifikant ansteigt. Somit wird deutlich, dass ökologische Finanzausgleiche eine dezentralisierende Wirkung auf Schutzgebietsausweisungen haben können. In beiden Fällen ist grundlegende institutionelle Voraussetzung für die beobachteten Auswirkungen, dass die Gemeinden entsprechende Naturschutzkompetenzen haben, also selbst Schutzgebiete ausweisen können.

Auf Grundlage dieser empirischen Erkenntnisse liefert die Arbeit drei Studien zu Politikgestaltung und institutionellem Design von ökologischen Finanzausgleichsystemen. Am Beispiel *Deutschlands* werden mittels einer institutionell und empirisch fundierten Politiksgestaltungsstudie die notwendigen Nachweise für einen strukturellen

Mehrbedarf der Bundesländer für Naturschutz erbracht und konkrete Vorschläge einer Integration in den föderalen Finanzausgleich entwickelt. Am Beispiel der *Europäischen Union* wird gezeigt, wie ökologische Indikatoren in den Verteilungsmechanismus von EU Finanzierungsprogrammen wie den Europäischen Fonds für regionale Entwicklung integriert werden könnten. Die Verteilungswirkung von Zuweisungen für das Ausmaß von Natura 2000 Gebieten und Habitatsqualität wird räumlich explizit modelliert. Ein Regressionsbaum zeigt, dass vor allem wirtschaftlich schwache Gebirgsregionen von solchen ökologischen Finanzzuweisungen profitieren würden. Somit ist der vorgeschlagene Allokationsmechanismus im Einklang mit den Kohäsionszielen des anvisierten EU Fonds. Am Beispiel eines hypothetischen Fonds im Rahmen der *Konvention über Biologische Vielfalt* werden verschiedene Gestaltungsoptionen entwickelt und deren Passfähigkeit in Bezug auf den jeweiligen Beitrag zur Erreichung international vereinbarter Biodiversitätsziele ermittelt. Die sozial-ökologische Gestaltungsoption setzt im Vergleich zu den ökozentrischen und anthropozentrischen Optionen die Anreize dort, wo die Fehlstelle zur Erreicherung international Biodiversitätsziele am größten ist.

Insgesamt zeigt die Dissertation theoretisch fundiert die Potenziale eines ökologischen Finanzausgleiches für fiskalische Anreizsetzung im Naturschutz auf, liefert entsprechende empirische Nachweise, und entwickelt drei konkrete Vorschläge für mögliche Anpassungen des Instrumentes auf föderaler, supra-nationaler und globaler Ebene.

Danksagung

Ich bedanke mich bei meinen Eltern, meinem Bruder, meiner Frau und meinen Freunden, die mich zu dem gemacht haben, der ich bin. Ich bedanke mich bei Irene Ring, Claudia Becker und Bernd Hansjürgens, die meine Dissertation betreut haben und mich auf meinem wissenschaftlichen Weg tatkräfig unterstützen. Ein Dankeschön an Christoph Schröter-Schlaack für die Disserationspatenschaft. Meinen Ko-AutorInnen, den Herausgeber*Innen und den anonymen Gutachter*Innen gilt mein Dank bezüglich der Publikationen. Ein besonderer Dank geht an meine GastgeberInnen in Rio de Janeiro, Lisabon und Quito für die Gastfreundschaft. Meinen Kolleg*Innen am UFZ und den Teilnehmern der Konferenzen, auf denen ich meine Arbeiten vorgestellt habe, ein Dankeschön für die hilfreichen Rückmeldungen. Dem Forum Ökonometrie und dem Mitteldeutschen Doktorandenprogramm Ökonomie mein Dank für die methodischen Diskussionen und Anregungen. Der Kaffeepause, dem Raum 009 und den Kellerkindern meinen Dank für die angenehme Atmosphäre. Es war eine gute Zeit!

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List of Abbreviations

ABS	access and benefit sharing
AR	autoregressive
BC	Budget Constraint
CBD	Convention on Biological Diversity
CF	EU Cohesion Fund
CI	Confidence Interval
CNUC	Cadastro Nacional de Unidades de Conservação
EARDF	European Agricultural Rural Development Fund
EFT	Ecological Fiscal Transfers
EMFF	European Maritime and Fisheries Fund
ERDF	European Fund for Regional Development
ESF	European Social Fund
EU	European Union
FAG	Financial Equalisation Act (Finanzausgleichsgesetz – FAG)
GDP	Gross Domestic Product
GMF	General Municipal Fund
HDI	Human Development Index
IBGE	Instituto Brasileiro de Geografia e Estatística
ICMS-E	Imposto sobre Circulação de Mercadorias e Serviços – Ecológico
ICNF	Instituto da Conservação da Natureza e das Florestas
IOER	Leibniz Institute of Ecological Urban and Regional Development
IPEA	Instituto de Pesquisa Econômica Aplicada
IUCN	International Union for Conservation of Nature
LIFE	EU Programme for Environment and Climate Action
MaßstG	Maßstäbegesetz – Standards Act
MCMC	Markov chain Monte Carlo
MMA	Ministério do Meio Ambiente
MRS	Marginal Rate of Substitution

N2k	European Natura 2000 Network
NGO	Non-governmental Organization
NUTS	Nomenclature des unités territoriales statistiques
PA	Protected Area
PES	Payments for Environmental Services
RNAP	Rede Nacional de Áreas Protegidas
SAC	Special Areas of Conservation
SD	Standard Deviation
SNUC	Sistema Nacional de Unidades de Conservação da Natureza
SPA	Special Protection Areas
UN	United Nations
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
USA	United States of America

For my love.

Inspirations

"Fiscal federalism is in vogue. Both in the industrialized and in the developing world, nations are turning to devolution to improve the performance of their public sectors. [...] The hope is that state and local governments, being closer to the people, will be more responsive to the particular preferences of their constituencies and will be able to find new and better ways to provide these services."

Wallace E. Oates (1999, p. 1120)

"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."

Irene Ring (2002, p. 418)

"Local governance executives (mayors) are more likely to support and invest in municipal natural resource governance when they perceive clear institutional incentives to do so, regardless of the degree of decentralization."

Krister P. Andersson und Elinor Ostrom (2008, p. 81)

"Governments, organizations, the private sector and financial institutions are encouraged to provide financial resources, including through new and innovative financial mechanisms, for the implementation of the [Nagoya] Protocol"

Elisa Morgera, Elsa Tioumani and Matthias Buck (2014, p. 332)

Preface

Somewhat surprisingly to myself, I have developed a sincere interest in studying fiscal constitutions by econometric means. Being a political scientist and sustainability economist by training, I want to understand potential avenues for sustainble development in terms of a just and environmental sound economic development. I am thus interested in tools and mechanisms that would allow us – as a global society – to transform unsustainable growth patterns to a world that harbours a diverse and rich life for most if not all of us.

In the search for instruments I encountered ecological fiscal transfers as a very promising instrument. It is promising in the sense that it changes existing fiscal transfer schemes by integrating ecological indicators regarding the provision of protected areas. It would thus not necessarily require additional funds but at the same time set incentives for the public provision of nature conservation and recognize the importance of such a public function within a system that has a heavy focus on economic development in terms of tax revenue generating activities. It balances the incentives currently inherent in these systems towards a more sustainable set of fiscal stimuli for local governments. Furthermore, the instrument is related to the organization of competencies within multi-level government structures and focuses on the lowest level governments which are – according to the idea of fiscal federalism – closer to the people, thus supposedly more apt for the fulfillment of local needs, and therefore interesting entities to look at. By balancing fiscal stimuli towards ecological public functions the designation of protected areas would not be coerced but compensated. The received revenue could still be spent on whatever purpose the municipality sees fit and still serve the need for conservation efforts.

Hence, I wanted to know how much potential the instruments actually hold and found econometric analyses the most convincing way of evaluating the effect ecological fiscal transfers may have on the designation of protected areas. To this end I chose to study the only two existing cases of a large scale implementation empirically: Brazil and Portugal. Moreover, I found this mechanism so promising that I had myself thinking about how the mechanism could be upscaled to other governmental levels. To that end I combined institutional analysis approaches with quantitative and spatially explicit analyses to assess the incentives and distributive patters of the simulated schemes.

The resulting cumulative thesis consists in the following five articles[1]:

- Droste, N., Lima, G.R., May, P.H., Ring, I. (2017) Municipal Responses to Ecological Fiscal Transfers in Brazil - a microeconometric panel data approach. *Environmental Policy and Governance* 27(4): 378–393. doi: 10.1002/eet.1760
- Droste, N., Becker, C., Ring, I., Santos, R. (2017) Decentralization effects in ecological fiscal transfers – the case of Portugal. *UFZ Discussion Paper 3/2017 submitted to Environmental and Resource Economics*.
- Droste, N., Ring, I., Schröter-Schlaack, C., Lenk, T. (2017) Integrating Ecological Indicators into Federal-State Fiscal Relations: A Policy Design Study for Germany. *Environmental Policy and Governance* 27(5): 484–499. doi: 10.1002/eet.1774
- Droste, N., Ring, I., Santos, R., Kettunen, M (2016) Ecological Fiscal Transfers in Europe – evidence-based design options of a transnational scheme. *UFZ Discussion Paper 10/2016 submitted to Ecological Economics.*
- Droste, N., Farley, J., Ring, I., May, P.H., Ricketts, T. (2017) Designing a global mechanism for intergovernmental biodiversity financing. *Currently a draft article in preparation for submission to Nature.*

I hope you will enjoy reading the thesis as much as I did developing it.

[1] The permissions that the publishers granted me to reprint the articles are gratefully acknowleged.

Part I

Introduction

Conservation and Fiscal Federalism

Biodiversity policy through the lense of public finance

Human activity on Earth has reached an unprecedented scale. A growing population, intensified production patterns and an increasing conversion of land have led to adverse effects on ecosystems and environmental health (MEA, 2005). The aggregate effects of this expansion threatens the very foundation for future development. Ecological carrying capacities are reaching their limits and some planetary boundaries have already been crossed. The loss in biodiversity is the topmost precarious dimension that imperils sustainable development and thus human livelihoods on the planet (Steffen et al., 2015). The decline in both functional and genetic diversity poses severe risks for the resilience of ecosystems and the ability to recover from shocks. Perturbations may thus have far larger impacts than up until now and large-scale ecosystems are being pushed to the brink of collapse. This situation calls for insurance policies in terms of biological conservation and ecosystem restoration.

(Inter-)national institution building is under way. Since the 1992 Earth summit in Rio de Janeiro, the United Nations have adopted conventions for the preservation of a healthy environment. Climate change, desertification and biological diversity are on the agenda. Their progress has led to the implementation of national and local policies for a sustainable development. Yet, these policies have not been able to halt the loss of biodiversity on a global scale. The Aichi biodiversity target indicators largely show an insufficient progess to reach the internationally agreed upon goals for biological conservation (CBD, 2014). Additional action is required. One of the main drivers for the loss of biodiversity is agricultural production. Sustainable solutions within agricultural land-use systems and integrated conservation measures are thus key to halt the decline in biodiversity (TEEB, 2015). Potential policy instruments include direct regulation through bans and land-use standards, setting incentives through economic instruments, and nudging behavioural responses through capacity building (TEEB, 2010). However, while often seen as the primary adressees, private land users are not the only relevant actors in this context (Vatn, 2015). The public finance school of thought provides the insight that public administrations' behaviour is a relevant factor for the use of natural resources (Boadway and Shah, 2009). The conservation policy mix is therefore not complete without instruments that address public agencies in their function to conserve nature (Ring and Barton, 2015).

Public finance provides a theoretical lense through which institutions and their behavioural effects for (local) governments can be identified and studied (Buchanan, 1967; Musgrave, 1959; Oates, 1972). At a principal and underlying level, (fiscal) constitutions organize the relation of various levels within (supra-)national goverment structures. They define each levels' competencies and shares of tax revenue. Figure 1.1 provides a schematic overview of several types of fiscal transfers that are in place in order to ensure sufficient fiscal capacity of various government levels for fulfilling their public functions. The role of the environment within such intergovernmental fiscal relations is still a relatively unexplored academic field and includes works on ecological public functions (Ring, 2002) and environmental fiscal federalism (Oates, 2005). Most of the works, however, deal with environmental pollution rather than conservation. At the same time, conservation policies are a wide-spread public function. Protected areas (PA) have been designated through legislative acts at least since the early 19th century.[2] Locally relevant ecological sites and culturally sacred sites have been protected for far longer (Berkes, 2008).

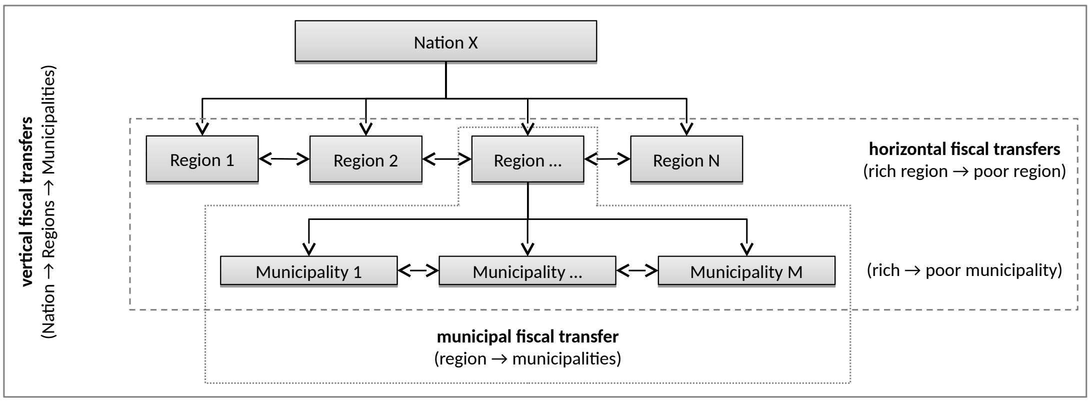

FIGURE 1.1: Overview of fiscal transfers in multi-level governments. *Source*: Droste et al. (2017b, p. 332)

This dissertation analyzes an innovative instrument with promising characteristics that may help to halt the loss of biodiversity through the lense of public finance. Ecological fiscal transfers (EFT) are an innovation from Brazil (Grieg-Gran, 2000; Loureiro, 2002; May et al., 2002; Ring, 2008c). After watershed protection to ensure freshwater supply for the capital of Paraná, Curitiba, the affected municipalities complained about their loss in (potential) tax revenue from land development and agricultural production (Grieg-Gran, 2000). The state government responded to the complaints by implementing a compensation scheme that assigned a share of the value-added tax revenue distributed among municipalities to the existence of PA in 1991 (Loureiro, 2002). The idea took hold in several other Brazilian states and has now been implemented in 17 out of

[2] Sustainable management practices in modern times go back to silvicultural needs for foresight once wood became a scarce resource and have been prominently publicized by John Evelyn (1664) and Hans Carl von Carlowitz (1732).

26 Brazilian states (The Nature Conservancy, 2014). In 2007, the instrument was implemented at the national level in Portugal (Santos et al., 2012). Proposals have been developed for a potential implementation in Switzerland, Germany, France, Poland, India, and Indonesia (Borie et al., 2014; Irawan, Tacconi, and Ring, 2014; Köllner, Schelske, and Seidl, 2002; Kumar and Managi, 2009; Mumbunan, 2011; Ring, 2008b; Schröter-Schlaack et al., 2014). The central innovative feature of EFT is that it ties fiscal transfers to the existence of PA. It thus constitutes a financial flow for public conservation action in terms of designated PA – which is not just a compensation payment but it is also seen as an incentive for the designation of additional protected areas (Loureiro, 2002; Ring, 2008c; Sauquet, Marchand, and Féres, 2014). EFT may thus help to counteract the loss in biodiversity through enhancing the provision of PA. Empirical analyses of such an incentive effect on the designation of protected areas are however largely lacking. Up to date there is no systematic review of whether the implementation of EFT leads to an increase in PA.

This dissertation aims at closing the gap and provide empirical evidence for the effect of EFT on PA. The corresponding research questions are:

1. Does the implemenation of EFT lead to an increase in PA?
2. Which institutional characteristics of the EFT schemes determine the outcome?
3. What policy advice can be drawn from the empirical analyses and quantitative modelling for the design and adaptation of EFT schemes?

In order to provide answers, the dissertation develops a microeconomic model in order to derive hypotheses that can then be tested through econometric techniques for studying the effect of existing EFT scheme in both Brazil and Portugal. Furthermore potential adaptations at federal, EU, and UN level will be evaluated based on quantitative simulations and econometric analyses. The structure of the cumulative dissertation is the following:

- Chapter 2 provides the theoretical background and a microeconomic model of the incentive effects in fiscal transfers in order to derive testable hypotheses and evaluative normative criteria for the subsequent analyses.
- Chapter 3 provides a microeconometric panel data analysis of the effects of the Brazilian EFT schemes on the designation of PA for data from 1991-2009 in order to assess the outcomes of introducing ecological criteria in intergovernmental fiscal relations.
- Chapter 4 provides a Bayesian structural time series analysis of the effect of the Portuguese EFT on the degree of centrality in conservation decisions for data from 1995 to 2014 and highlights essential institutional features for a decentralizing effect.

- Chapter 5 adapts the idea of municipal EFT to a federal to state level fiscal relations setting and provides a three step policy analysis regarding institutional framework, empirics of conservation patterns, and a quantitative simulation of EFT in the German fiscal equalization scheme.
- Chapter 6 provides a policy proposal for an adaptation of EFT to the EU network of protected areas and the distribution of EU funds. The spatially explicit quantitative simulations are assessed with a regression tree in terms of allocative patterns and socio-economic characteristics of potential beneficiaries.
- Chapter 7 contributes a proposal for a global intergovernmental mechanism for financing biodiversity conservation. Three different design options, ecocentric, socio-ecological and anthropocentric are developed and assessed in terms of the fitness of the resulting financial incentives with regard to reaching international biodiversity targets.
- Chapter 8 concludes with summarizing overall findings and remarks on lessons learnt and value added.

For all analyses the source code and the datasets of the econometric analysis in the **R** environment are provided through links to a personal github repository in appendix B such that the findings are fully reproducible. Thereby, this dissertation contributes a systematic and reproducible set of analyses regarding the effect of existing and evidence based policy designs of proposed EFT schemes. The thesis covers a range from sub-national to international level intergovernmental fiscal relations with respect to institutional features that may incentivize public nature conservation efforts and thus help to halt or slow down the loss of biodiversity.

Part II

Theoretical Model

Decisions and Incentives

Local Provision of Public Goods and Intergovernmental Fiscal Relations

Abstract: This chapter introduces a public finance perspective on the organisation of multi-level government structures and mechanisms. The analysis is based on two main theoretical propositions: a) given differences in both local preferences and costs of provision, decentralization and competition between municipalities allow for efficient satisfaction of preferences in each locality; b) given spatial spillover effects between jurisdictions, compensating for the spillover benefits through corresponding fiscal transfers would set local provision levels closer to societally desirable output levels. In order to derive hypotheses for the empirical parts of this dissertation, I develop a microeconomic model about decentral decision makers' responses to fiscal transfer scheme incentives. The basic model shows the behaviour of a boundedly rational decision maker under budget constraints where fiscal transfers may enhance available budgets and change relative costs such that there is a greater supply of the public good in question. A model extension with a two municipality competition for performance-oriented fiscal transfers clarifies the conditions under which the provision of the good in question is enhanced and fiscal equalization occurs. Furthemore, normative criteria for the evaluation of potential outcomes are discussed.

2.1 Spillover Benefits and Intergovernmental Fiscal Relations

The analysis of local decision making starts from a basic setting of differences between decentral jurisdictions. Regarding the local provision of public goods, the goods' properties and the related free-rider problem will be introduced. Potential solutions to free-riding originate from the theoretical realm of fiscal federalism and a public finance perspective on the organisation of intergovernmental fiscal relations. The relevant proposals focus on two elements: municipal competition and fiscal transfers.

Setting the scene: suppose there is a range of local jurisdictions that are decentral authorities of a multi-level government such as a federation or a unitary government with multiple jurisdictional levels. These local jurisdictions may have different constituencies, thus different preferences, and different resource endowments, thus different resource prices. The differences in relative resource prices and rates of marginal substitution may thus result in varying levels of provision among the local jurisdictions. While such differences may constitute problems for the provision of public goods, there are proposals of how to deal with and make optimal use of such differences. I will introduce both problems and solutions in the following.

Samuelson (1954, 1955) proposed two types of goods. Pure private goods are a) rivalrous in the sense that once they are consumed they can no longer be enjoyed by another person, and b) they are excludable in the sense that a person can be denied access to them. Pure public goods are a) nonrivalrous, such that a consumption by one persons does not inhibit the consumption by others, and b) nonexcludable, such that everyone may enjoy them (ibid.). Buchanan (1965) added a third type, so called club goods. These may be provided for small groups of people who may all enjoy them unrivalrously but they may exclude non-members from doing so. Ostrom and Ostrom (1977) proposed a fourth type, so called common-pool resources. These are rivalrous and nonexcludable.

For goods that are nonexcludable, i.e. public goods and common-pool resources, the *free-riding problem* states that those who benefit from it cannot easily be obliged to pay for them, costs of provision can thus not necessarily be covered. This in turn results in an underprovision (Baumol, 1952; Hardin, 1968; Olson, 1965).[3] There are different solutions available, most of them deal with specifying property right institutions. Governments may regulate the use of public goods, for example through *bans or taxes* (Baumol, 1952; Pigou, 1920). Coase (1960) has shown that, irrespective of the initial distribution of property rights, a *bargaining* solution between land-users could also lead to an optimal solution – if there were no transaction costs involved. Common property regimes may govern the resource use through *institution building*, defining boundaries, specifying provisioning rules, arranging participation in collective choices, installing monitoring systems, implementing sanctioning rules and the like (Ostrom, 1990). In the following, the analysis will focus on free-rider problem solutions for nonexcludable goods and services that are provided by public entities, such as local jurisdictions.

Based on the idea that different local jurisdiction may provide different levels of goods and services, Tiebout (1956) formulated a *voter migration* solution to free rider problems. Tiebout proposed competition between local jurisdictions as a means to achieve optimal tax rates and public good provision for each jurisdiction. Assuming that people can freely change their place of living, voter migration would yield the optimal community of people for a jurisdiction since local governments would adjust tax rates and public good provision until preferences are best satisfied and everything settles into equilibrium. People would live where they would be willing to pay for the provided public goods and free-riding would thus be minimized. Similarly to the effect of transaction costs in the Coase bargaining solution, introducing costs of movement for citizens might however change the model outcome (Tiebout, 1956). While migration might thus not always be a feasible solution, the important insight is that decentralized decision making may yield welfare gains if regulatory competition is allowed between local jurisdictions. Uniform solutions may not sufficiently take into account differences in local preferences and costs of provision. For example, a uniform tax rate may be too

[3] The free-riding problem of collective action is based on the assumption of individually rational behaviour, where actors only maximize their individual net pay-off, e.g. by minimizing their own costs through unsanctioned free-riding. This in the end leads to collective irrationality (Olson, 1965).

high for the preferences of one local jurisdiction's population but too low for another jurisdiction. Locally adjustable tax rates may take into account such differences.[4]

Spatial spillover benefits constitute another important dimension of the free-riding problem. Roads do not stop at jurisdictional borders, people's education and scientific discoveries may be used elsewhere, or a clean river may have downstream beneficiaries. Olson (1969) proposed a 'principle of fiscal equivalence', stating that those who benefit from a public good should also have to pay for its provision. This would yield an optimal level of supply due to a (spatial) match between demand and supply. Regarding a local public good where only the local population receives benefits of it, the local government will be the best suited provider. In case it is a public good from which the whole nation benefits, the national government should provide it. But Olson also takes the supply side into account. When a higher government level constituency benefits from a particular public good, but diseconomies of scale call for local provision, *fiscal transfers* from higher to lower government levels may ensure optimal supply at minimal costs (Olson, 1969). The good in question can be provided cheaper at local levels but the potential benefits to the overall society call for a corresponding compensation from a higher level government such that local level providers internalize those benefits in their behavioral rationale. As a result societally desirable levels are obtained. Another case for a fiscal transfer to the local level is when there are spatial spillover benefits of the good in question and neighboring jurisdictions would also benefit from enhanced supply (Olson, 1969). The important insight is that fiscal transfers between government levels (both horizontally and vertically) may help to align local provision with overall societal interests, when there are differences in marginal costs of provision which could be of use or when spatial spillover effects affect other jurisdictions.

Building upon such insights, fiscal federalism and public finance approaches analyze effects of the organisation of competencies in multi-level government systems, and corresponding governmental activity such as (de)central regulation, taxing, or the (de)central provision of public goods (Boadway and Shah, 2009; Brennan and Buchanan, 1980; Musgrave, 1959; Oates, 1972, 1999; Zimmermann, Henke, and Broer, 2011). This can be done in both in positivistic terms (section 2.2), and in normative terms (section 2.3). The first generation fiscal federalism often assumes a welfare maximizing governmental behavior (Brennan and Buchanan, 1980; Feld, 2014). This assumption may not fully be realistic. Local decision makers may seek their own (constituency's) benefit, not the larger society's one. The second generation fiscal federalism therefore moves to analyze incentives inherent in intergovernmental fiscal relations and the effects on local

[4] There can be benefits to interjurisdictional competition and decentralization (Faguet, 2004; Oates and Schwab, 1988; Rubinchik-Pessach, 2005; Sorens, 2014). This is not to say that such competition is always a solution or even preferable (Fischer and Wigger, 2016; Sinn, 1990). Some constitutions do not allow for much taxing competition between (lower) government levels. Some public goods or services may be provided more efficiently at a central level due to economies of scale. Some uniform taxes may avoid a race to the bottom. The main point is that when there are elements of such a competition, their effects should be analyzed, see subsection 2.2.2.

decisions (Oates, 2005; Weingast, 2009). I will follow this line of analysis and analyze incentive effects in fiscal transfers.[5]

For such an analysis it is necessary to differentiate between the different available types of transfers (Boadway and Shah, 2009). *General-purpose transfers* constitute unconditional budget support for lower government levels such that they may fulfill their general public functions. They increase the spending capacity while allowing for greatest local spending autonomy. *Specific-purpose transfers* have spending conditions that earmark fiscal support for specific task and programme expenditures. They are designed to create incentives for lower government levels in the form that they lower costs of provision. They may have matching or co-financing requirements where both higher and lower government levels contribute funds. These transfers allow only for a limited local spending autonomy. *Performance-oriented transfers* are conditioned on the provision of a particular output (Boadway and Shah, 2009). Similar to specific-purpose transfers, they provide incentives by lowering the costs for a particular task since they are payed for outputs of that task. At the same time, they allow for a greater local spending autonomy since the obtained transfers can be spent in any way the local jurisdiction sees fit. They constitute a hybrid form between general and specific-purpose transfers: they attach the transfer to a particular performance but do not come with further spending strings (cf. chapter 6). EFT are performance-oriented transfers, since they are based on the existence (and quality) of protected areas within the receiving jurisdiction's territory but have no pre-defined spending purposes. The following model will hence focus on fiscal transfers from a higher level government to municipalities that are performance-oriented and may be spent upon all possible policy areas.

2.2 A Microeconomic Model of Incentives in Fiscal Transfers

Parts of the following microeconomic model have been employed in later chapters (see sections 3.4 and 4.3 but here it will be formulated for the first time in its entirety. The model is a reformulation of the basic model presented in Boadway and Shah (2009, chapter 9), is furthermore based on the structure of consumer choice models (Varian, 2010, chapter 8), and is extended by my own municipal competition modelling. Section 2.2.1 introduces the basic model of a boundedly rational decision maker's behaviour under budget constraints and the effect of fiscal transfers on relative costs. Section 2.2.2 extends this model to a situation where two municipalities compete for the funds available through a performance-oriented fiscal transfer.

[5] This is a perspective similar to the one taken by new institiutional economics, since it analyzes the effect of institutions on behaviour (Coase, 1937; Coase, 1960; North, 1990; Ostrom, 1990; Williamson, 1996).

2.2.1 Unlimited Wants and Resource Constraint Decisions

The analysis in this section presumes a methodologically individualistic perspective by conceptualizing decentral decision making as if there was a single decision maker involved. Such a local decision maker may aim to fullfil the whishes of his constituency plus personal interests.[6] Such a conglomerate of collective and individual preferences could be understood as a decisional basis – a preference set – for a local politician.[7] This preference set can be understood as a list of projects and policies the local decision maker aims to realize in order to provide the constituency with the desired (public) goods and services. In theory, the decision maker may want to realize all of these preferences to the maximum extent irrespective of their ranking. This could be called a situation of *unlimited wants and needs*.[8] But we live in a world of finite resources. Thus, there are limits to what a local decision maker can achieve. There are several levels of constraints that may limit the satisfaction of preferences for our hypothetical individual decision maker, such as limited knowledge, bounded rationality, organisational or administrative capacities, market conditions, jurisdictional competencies, or resource system boundaries (cf. Daly and Farley, 2010; Kahneman and Tversky, 1979; Sen, 1977). That is to say, simultaneously with hypothetically unlimited wants the decision maker faces a situation of *limited available resources*, where resources are meant in a broad sense such as financial, human and natural capital. In line with one of the main objective of the dissertation, the effect of EFT on protected area, the analysis will focus on one of the limits that is quite clearly applicable in the context of local decision making: *budgetary constraints*. Local administrations only have a certain amount of financial resources that they may spend on the realization of all their preferences.

Here is where the model starts. Let M denote the available monetary budget of the local decision maker that may be spent on the provision of a bundle of public goods and services. Limiting the problem to two dimensions, the bundle may have two policy fields, say X, for conservation action, and Y for a composite of all other policies. Given that the policies have costs (or prices) to be implemented, let those be denoted by c_x and c_y, correspondingly.

Assuming that all available budget will be spent, this can be formualated as

[6] This is a simplification of several works on (local) political decision making (Black, 1958; Buchanan, 1967; Downs, 1957; Hansjürgens, 2000; Krueger, 1974; Lindahl, 1919; Musgrave, 1959; Samuelson, 1954; Simon, 1955; Wicksell, 1889). There may be a variety of both intrinsic motivations and external factors influencing local decision making - among the latter incentives inherent in fiscal transfer can be found.

[7] The local constituency may consist of several people with different familiy backgrounds, different educational levels, different jobs and different personalities. This, in turn, can result in diverse and potentially unstable aggregated societal preferences of the constituency (Arrow, 1963; Buchanan and Tullock, 1962; Olson, 1965). But given that local decision makers have to decide, and for the sake of simplicity, I will assume that both the voters' preferences, thus local welfare, and some self-interest for being re-elected are interdependent arguments of the local decision maker's utility function.

[8] This is based on an assumption of insatiable preferences or monotonicity of utility curves. There may however be optimal levels of consumptions beyond which the net benefit decreases due to increases of "costs" of consumption. Depending on the type of good in question "the more the better" may not necessarily hold when there are side-effects to consumption – think of socially accepted intoxicants.

$$M = c_x X + c_y Y \ . \tag{2.1}$$

This budget constraint does not alone define, which combination of policies will be chosen. Decisions on the amount of both policies will also be determined by their relative importance for the local constituency. Let U denote the utility, that the local decision maker obtains from both personal interest in policies Y and X and their contribution to preference satisfaction within the constituency.[9]

Solving the maximization problem

$$\max_{X,Y} \quad U(X,Y) \tag{2.2}$$
$$\text{s.t.} \quad M = c_x \mathrm{X} + c_y \mathrm{Y} \tag{2.3}$$

via a Langrangian function with its multiplier λ,

$$L = U(X,Y) + \lambda[M - c_x X - c_y Y] \ , \tag{2.4}$$

and setting its partial derivatives with respect to Y, X and λ to 0, yields the first order conditions

$$\frac{\delta U}{\delta X} - \lambda c_x = 0 \ , \tag{2.5}$$
$$\frac{\delta U}{\delta Y} - \lambda c_y = 0 \ , \tag{2.6}$$
$$M - c_x X - c_y Y = 0 \ , \tag{2.7}$$

where both equations (2.5) and (2.6) can be solved for λ such that

$$\lambda = \frac{\frac{\delta U}{\delta X}}{c_x} = \frac{\frac{\delta U}{\delta Y}}{c_y} \ . \tag{2.8}$$

The optimal choice between the two policies would thus be where the marginal rate of substitution ($\frac{\delta U}{\delta X} / \frac{\delta U}{\delta Y}$) equals the cost ratio, since rearranging (2.8) yields

$$\frac{\frac{\delta U}{\delta X}}{\frac{\delta U}{\delta Y}} = \frac{c_x}{c_y} \ . \tag{2.9}$$

[9] A more precise formulation of the maximization problem would be to assume that the policies X and Y produce corresponding public goods, for example with production functions $f(X) = E$, for environmental quality, and $g(Y) = D$, for composite development policies, and a resulting utility function of $U(E,D)$. This would specify that the benefits are obtained from outcomes of the policies not the policies itself. If the budget constraint (equation 2.3) would simultaneously be formulated as $M = c_{f(X)} f(X) + c_{f(Y)} Y$, the overall problem could be stated as $\max_{E,D}$ s.t. $M = c_e E + c_d D$ and the result would still hold.

In spite of being simplistic, important insights can be drawn from the model. If budget constraints are binding, the total outcome will be defined by the available income (see equation 2.1). More importantly, however, the model tells us that the cost ratio determines how much will be spent on the different policies (see equation 2.9). In order to implement corresponding public functions, the relative costs of the policies determine the composition of the resulting goods and services the local jurisdiction can provide. This is to say that *changes in relative cost will yield differences in how much of different policies can be expected*.

This model can furthermore be used to determine the demand for a particular policy. If one assumes a diminishing marginal rate of substitution (*MRS*) between the policies[10], the resulting utility function could be formulated as Cobb-Douglas function

$$U(X,Y) = X^{\alpha}Y^{1-\alpha} \tag{2.10}$$

where α is the fraction of income used for X and $1-\alpha$ the income fraction used for Y. In the following I will employ a monotonic transformation of the Cobb-Douglas function, $U(X,Y) = \alpha \ln X + (1-\alpha)\ln Y$ (Varian, 2010, p. 111 ff.). Maximizing this utilitiy function with respect to Y and X with the budget constraint (2.3), yields the first order conditions

$$\frac{\alpha}{X} - \lambda c_x = 0\,, \tag{2.11}$$

$$\frac{1-\alpha}{Y} - \lambda c_y = 0\,, \tag{2.12}$$

$$M - c_x X - c_y Y = 0\,. \tag{2.13}$$

Solving equations (2.11) and (2.12) for λ gives

$$\frac{\alpha}{c_x X} = \frac{1-\alpha}{c_y Y}\,. \tag{2.14}$$

Crossmultiplying (2.14) yields

$$\alpha c_y Y = c_x X - \alpha c_x X\,. \tag{2.15}$$

which in turn can be solved for c_yY, giving $c_yY = \frac{c_xX}{\alpha} - c_xX$. Inserting this into the budget constraint first order condition (2.13), c_xX cancels out in $M - c_xX - \frac{c_xX}{\alpha} + c_xX = 0$. Rearranging yields

$$\alpha M = c_x X \tag{2.16}$$

such that we arrive at a definition of the Marshallian demand for policy X (Varian, 1992)

[10] A dimininishing MRS means that the less there is of a good in question the higher becomes the trade-off in terms of the other.

$$X = \frac{\alpha M}{c_x} \, . \tag{2.17}$$

Hence, the demand for conservation policies X depends on the ratio of available budget M, its cost c_x, and a factor of its marginal utility, α. This is to say, the larger the budget, the greater the demand for the good, and the smaller the cost, the greater the demand. Such is the case for normal goods. But it also implies that the degree of these movements will depend on the parameter α, the income fraction used for X.

Now, the effect of an introduction of a fiscal transfer on expenditures for public goods can be determined. Suppose that the transfer is financed through higher government level sources that do not change the lower level government's budget in any other way than the transfer itself (an assumption I will later on relax). Then, the first effect of a performance-oriented fiscal transfer is rather simple. Since the transfer can be spent on whatever purpose it increases the available budget, such that $M' > M$ and greater quantities of X and / or Y can be afforded. For illustrative purposes, I will now make use of a graphical analysis. Figure 2.1 shows the budget increase as an outward shift of the budget constraint BC from BC_1 to BC_2.[11] As a result Y_1 increases to Y_2, and X_1 to X_2. The actual increase in quantities lastly depends on the slope of the utility indifference curve ($I = U(X, Y)$), for a Cobb-Douglas utility function on the parameter α (see equation 2.17). The second effect of a performance-oriented fiscal transfer is a bit more complex. Through the per unit output transfers policy X becomes cheaper such that the new cost $c'_x < c_x$.[12] This implies a relative cost change and the slope of the BC becomes less steep, $c'_x/c_y < c_x/c_y$. Figure 2.1 represents this as the shift from BC_2 to BC_3.

Consider the demand for X as a function of cost c_x and available budget M (see equation 2.17), then the total effect of a cost change from c to c' can be written as the difference between the demand for X at old and new costs (Varian, 2010, chapter 8).

$$\Delta X = X(c'_x, M) - X(c_x, M) \tag{2.18}$$

The total effect of the relative cost change can be decomposed with the Slutsky equation into a substitution effect and an income effect (Varian, 2010, chapter 8)[13]. The *Slutsky substitution effect* results from a change in relative costs, if the budget M would simultaneously change to M^s such that the old good bundle in point B can still be afforded. Figure 2.1 displays this pivot as the grey dotted budget constraint going

[11] The buget constraint as depicted can be understood as a reformulation of equation (2.1) to $Y = M/c_y - (c_x/c_y)X$, where M/c_y is the intercept and $-c_x/c_y$ is the slope.

[12] The cheaper costs can be formalized with a cost function derived from a profit function. Let $\Pi = pq - cq$ denote a profit function of a policy, with q quantities of output, p income generated per unit, and c unit costs. A performance-oriented transfer yields an additional transfer T per unit, such that $\Pi' = \Pi + Tq$. Since $\Pi' > \Pi$, $c' = p - \Pi'/q < c = p - \Pi/q$.

[13] I will call these effects Slutsky substitution effect and Slutsky income effect, to avoid confusion with the pure income effect where there is an ordinary budget increase. For a formal decomposition see appendix A.

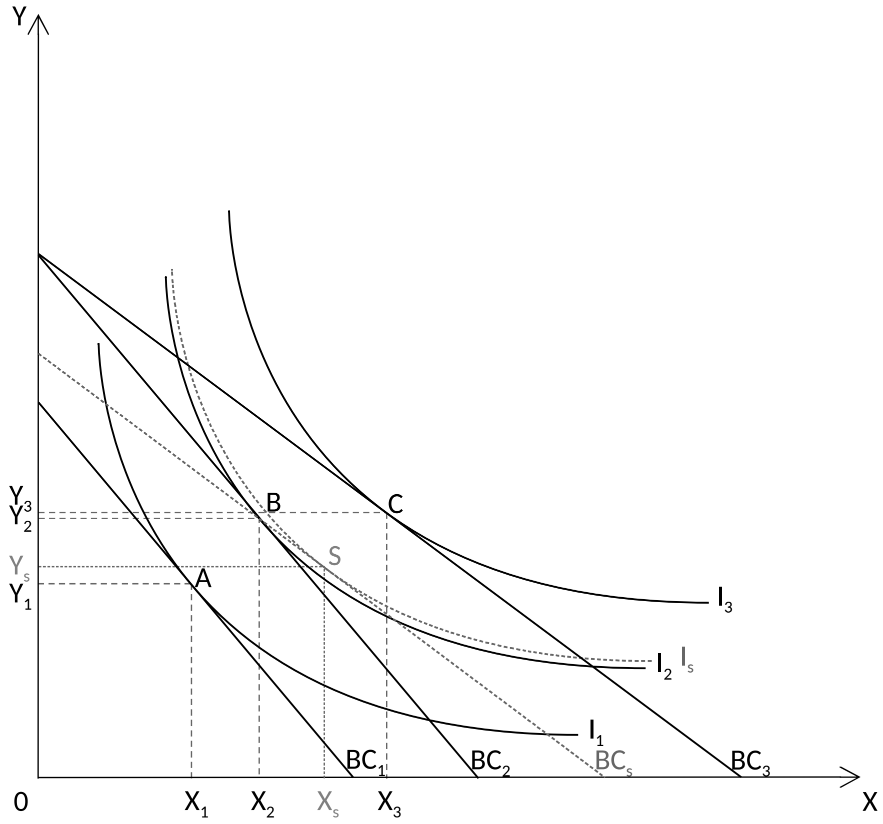

FIGURE 2.1: Local government spending behavior under budget contraints after introduction of fiscal transfers. *Source*: author's work based on Boadway and Shah (2009, chapter 9) and modified from Droste et al. (2017c)

through B. Through the pivot a new indifference curve, I_s can be obtained. The substitution effect is a hypothetical movement which would result in X_s and Y_s. It shows how much a decision maker would have to substitute X with Y to obtain the new utility level in point S, in this case $Y_2 - Y_s$ for $X_s - X_2$ under a simultaneous change of M to M^S. Formally, the Slutsky sustitution effect ΔX^S according to Varian (2010, chapter 8) is

$$\Delta X^S = X(c'_x, M^S) - X(c, M) . \tag{2.19}$$

But a relative cost change also results in an *Slutsky income effect*. Since X becomes cheaper, this frees up income which can be spend on whatever good, say the shift of the grey dotted line BC_s to BC_3 resulting in X_3 and Y_3. According to Varian (2010, chapter 8) the Slutsky income effect ΔX^N can be formulated as

$$\Delta X^N = X(c'_x, M) - X(c'_x, M^S) . \tag{2.20}$$

where c_x' is the new cost, and M^S is the Slutsky adjusted income used for the calculation of the Slutsky substitution effect, and $M - M^S$ is the freed up income.

It is worth noting that X is depicted as a normal good in figure 2.1, which means a lower cost will result in a greater demand. Futhermore, figure 2.1 shows a situation where some of the freed up income would also be spent on good Y and those in the constituency who would prefer Y over X would be better off in point C even when compared to B. This, however, ultimately depends on the size of the cost changes and again the actual shape of the indifference curve, and is thus not easily generalizable.

When a performance-oriented fiscal transfer is financed from other, higher government level resources and there are no further effects on the local jurisdiction's budget, the hypothesis derived from this model is nevertheless quite clear. There will be two effects on X:

i) a greater income, shifting the budget constraint outwards, and increasing the demand for X (the pure income effect), and

ii) a cheaper policy cost, increasing the demand for X further (the relative cost change, decomposed into the Slutsky substitution effect plus the Slutsky income effect).

Hence, under the assumption of a normal good and no further buget changes than through the transfer, I formulate

Hypothesis 1 *A performance-oriented fiscal transfer increases public spending on the policy in question and thus increases corresponding outputs.*[14]

Let me now relax the assumption that the introduction of a fiscal transfer does not affect the available budget. In a situation where the introduction of a performance-oriented fiscal transfer is a change in the distributive mechanism for a part of the fund but the overall fund size remains the same. This would constitute a situation where not A but B is the status quo in figure 2.1 - there will be no increase in budgets. The performance-oriented transfer would still cause a relative cost change and thus a pivot from BC_2 to BC_3. Even under such a scenario, hypothesis 1 would remain the same, although the effect would likely be smaller, for example $X_3 - X_2$ instead of $X_3 - X_1$ in figure 2.1.

2.2.2 Competition for Performance-Oriented Fiscal Transfers

As a final modelling step, let me introduce municipal competition for obtaining the funds. Consider the following hypothetical situation:

i) There are two municipalities. One, A, has a low budget but largely unoccupied nature, low levels of economic development, and little infrastructure such that nature

[14] Hypothesis 1 will be employed in both the article on Brazil (chapter 3) and the article on Portugal (chapter 4), asking: *Does the introduction of EFT increase (municipal) protected areas?*.

conservation is cheap but the composite other policies are relatively expensive. A large national park has been designated. The other, B has a higher budget, high economic development, and good infrastructure, but little wilderness such that there are low costs for the composite policies but conservation is relatively expensive due to high opportunity costs among conflicting land use interests. There are some but small scale conservation areas.

ii) There is a status quo fiscal transfer fund that co-determines available municipal budgets beyond municipal tax income. A performance-based fiscal transfer, paid per output unit, is introduced for a portion of the overall fiscal transfer fund but the overall funds remain the same. This constitutes a situation where the two municipalities would have to compete for the fund portion that is now only available through the performance-oriented transfer.

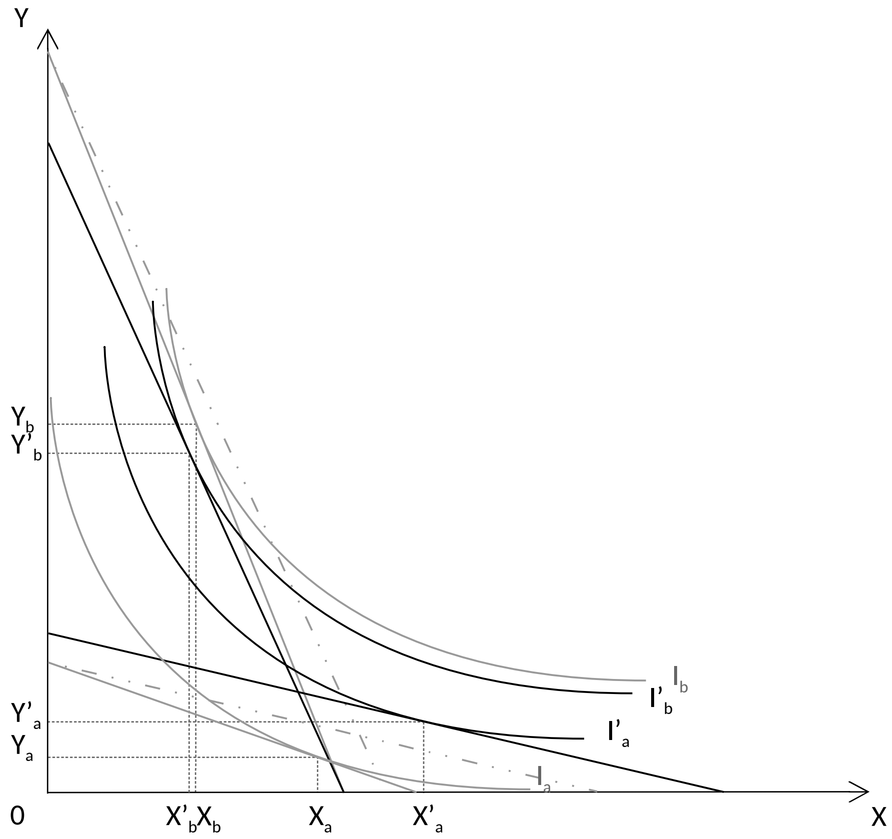

FIGURE 2.2: Local government competition for performance-oriented fiscal transfers. *Source*: author's work

Figure 2.2 provides some intution about the potential outcome on conservation policies X. It depicts two important results that may be obtained under conditions I will later derive: i) the gain in X_a can outweigh the loss in X_b, such that there would be more overall conservation action, ii) the indifference curves can get closer to each other (I'_a and I'_b in comparison to I_a and I_b) such that fiscal equalization may be obtained. This is due to two effects: the change in relative costs such that conservation action gets cheaper for everyone, including the substitution and the income effect (represented as a pivot of the grey to the grey dashed and dotted budget constraints); and the pure income effect resulting from the introduced competition where B loses while A gains (represented by the shift of the grey dashed and dotted budget constraints to the black ones).

From section 2.2.1 we already know that the effect of the relative cost changes in this model world is positive for both municipalities as long as we assume X to be a normal good. If there was no competition for the fund, the outcome would be like hypothesis 1 in both municipalities. The remaining outcomes are due to the redistributive changes due to competition that affect available budgets for both municipalities.

Per definition of the competitive situation conservation is cheaper in A. Hence,

$$c_{x_a} < c_{x_b} \, . \tag{2.21}$$

However, the per unit transfer is equal for both municipalities. Everyone gets the same transfer for a unit of conservation area such that the cost differences are the same.[15]

$$c'_{x_a} - c_{x_a} = c'_{x_b} - c_{x_b} \tag{2.22}$$

We also know that the overall fund size remains constant such that it is a zero-sum outcome, and the gain of one is the loss of the other. Therefore

$$\Delta M_a = -\Delta M_b \, . \tag{2.23}$$

where $\Delta M_a > 0$ due to the larger initial endowment with protected areas. Furthermore, the effect of a pure income change induced by the competition on X in municipality i is with equation (2.18)

$$\Delta X_i = X_i(c'_{x_i}, M'_i) - X_i(c'_{x_i}, M_i) \, , \tag{2.24}$$

where c'_i is the new cost after the relative cost change and M'_i is the new budget after competition. We can insert the Marshallian demand function $X = \alpha M / c_x$ (see equation 2.17), such that

$$\Delta X_i = \frac{\alpha_i M'_i}{c'_{x_i}} - \frac{\alpha_i M_i}{c'_{x_i}} = \frac{\alpha_i \Delta M_i}{c'_{x_i}} \, . \tag{2.25}$$

[15] This is a simplifying assumption, since the per unit transfers may depend on the performance of both municipalities and the overall fund size, e.g. if transfers $T_i = P_i F / \sum_{j=1}^{n} P_j$ are the share of the output unit performance P of municipality i of the fund F that is distributed among all n municipalities according to their performance. The addditional transfer for an increase of k units in municipality i would therefore be $\Delta T_i = (P_i + k_i)(F/(\sum_{j=1}^{n} P_j + k_i)) - P_i(F/\sum_{j=1}^{n} P_j)$ – which is not even symmetric in the two municipalities case but depends on the proportion of performances. Model results hold as long as the T_i does not change the relation $c_{x_a} < c_{x_b}$

where $\Delta M_i = M_i' - M_i$ is the change in budget.

The total effect of the combined pure income effect in both municipalities A and B can thus be formulated as

$$\begin{aligned} \Delta X &= \Delta X_a + \Delta X_b \\ &= \frac{\alpha_a \Delta M_a}{c'_{x_a}} + \frac{\alpha_b \Delta M_b}{c'_{x_b}} \end{aligned} \tag{2.26}$$

We know by equation (2.23) that $\Delta M_a = -\Delta M_b$. Given that $\Delta M_a > 0$, it follows that

$$\Delta X > 0 \iff \frac{\alpha_a}{c'_{x_a}} > \frac{\alpha_b}{c'_{x_b}}. \tag{2.27}$$

Whether the competition increases overall supply of X thus hinges on the the ratios α_i / c'_{x_i}. In order to compare cost ratios to preference ratios, we can rearrange the right-hand side inequality to

$$\frac{\alpha_a}{\alpha_b} > \frac{c'_{x_a}}{c'_{x_b}}. \tag{2.28}$$

This means that the ratio of costs must be smaller than the ratio of income fractions spent on X_i if competition is to increase overall X. We can simplify further. Since we know from equations (2.21) and (2.22) that $c'_{x_a} < c'_{x_b}$, we can deduce that for all $\alpha_a \geq \alpha_b$, $\Delta X > 0$. This is to say, whether competition increases overall X depends on relative preferences in the municipalities. If the preference for conservation policy X_i in the municipality where it is cheaper is at least as strong as in the other, we can safely assume that the competition increases X. I therefore hypothesize

Hypothesis 2 *If the muncipality where the policy in question is cheaper prefers that policy at least as strongly as elsewhere, a performance-oriented fiscal transfer induced competition between municipalities increases the provision of that policy and its outcomes.*[16]

We have now seen that under a competitive situation the outcome of total X depends on the relation of preferences and costs across municipalities, but that it does not depend on the initial distribution of income. However, if the competition is introduced through a change in budget allocation mechanisms without an overall budget increase it will have distributional effects – which are yet to be explored.

By definition of the scenario,

$$X_a > X_b \,, \tag{2.29}$$

and

$$M_a < M_b \,. \tag{2.30}$$

[16] The assessment of preferences goes beyond the scope of this dissertation and will remain for future research.

Under these conditions $M'_a > M_a$ and $M'_b < M_b$.[17] The distance between budgets M_a and M_b will thus decrease and a performance based fiscal transfer induced competition results fiscal equalization.[18] I therefore hypothesize

Hypothesis 3 *If the poorer muncipality has a higher initial output of the policy in question the introduction of performance-oriented fiscal transfer competition results in horizontal equalisation in terms of fiscal capacities.*[19]

2.3 Evaluating Outcomes from a Normative Perspective

Fiscal policies can be assessed from both positivistic and normative perspectives. The hypotheses derived from theoretical modeling of performance-oriented fiscal transfers will be employed in the empirical and thus positivistic analysis in the subsequent chapters, i.e. with regard to effects of an EFT introduction in Brazil and Portugal. The normative dimension has yet to be developed in order to provide terms against which to evaluate the outcomes of EFT.

The canonical dimensions of public finance serve as guiding principles to evaluate fiscal policy outcomes (Boadway and Shah, 2009; Buchanan, 1950; Feld, 2014; Hansjürgens, 2000; Musgrave, 1969; Zimmermann, Henke, and Broer, 2011). The first, economic efficiency, is about an efficient allocation of resources in the sense that they are employed where greatest net benefits may be obtained. The second, distribution of income, is about fiscal imbalances and equalizing measures to reduce mismatches between fiscal needs and fiscal capacities such that public functions can be met for the whole of a population. The third, macroeconomic stability, may not directly apply in the case of biodiversity protection, but if it was called systemic resilience instead, conservation would be a part of that dimensions. It would thus be about increasing the resilience of a multi-level government system to shocks and risks among various levels, such as economic and environmental ones. I will shortly introduce these criteria and relate them to the hypotheses derived in section 2.2.

In terms of *economic efficiency*, there are two basic factors that determine the net benefit of a policy outcome, costs and benefits, and the related problems of free riding on spillover benefits and corresponding underprovision. A performance-oriented fiscal transfer would thus increase economic efficiency if it would incentivize a) provision

[17] These assumption are somewhat realistic since large conservation areas are often realized where there are areas of relatively undisturbed nature available. These are most likely not highly developed, densely populated areas. But it is not necessarily always the case and remains an empirical question.

[18] It should be noted that these are effects of an introduction of a performance-oriented fiscal transfer and thus static. A dynamic analysis of strategic behaviour among these municipalities once the performance-oriented fiscal transfers are introduced is beyond the scope of this dissertaion but poses an interesting question for future research.

[19] This hypothesis will be analyzed in chapter 6 where the potential beneficiaries of a European EFT scheme policy proposal are identified and their economic characteristics are assessed, asking: *Cui bono?*. Furthermore, the empirical analysis in chapter 5 shows that per capita nature conservation area in Germany is significantly correlated with GDP per capita.

where it is cheapest, b) provision where it is most beneficial in gross terms, and c) increased provision of an underprovided public good. As we have seen in the theoretical analysis, performance-oriented transfers hypothetically create an incentive to provide a larger amount of the policy in question. If the empirical analyses find that this is the case, there will be an evaluative basis to conclude an increase in economic efficiency – given that the policy in question is underprovided. The question that remains is whether the incentive leads to an increase where it provides the greatest *net* benefits which relates to both costs and benefits. If a performance-oriented fiscal transfer provides an equal incentive per output unit those decentral governments with the lowest costs of provision will be most incentivized to increase provision since their net income from the transfer will be largest. This, however, does not necessarily ensure that the increase occurs where it provides the largest net benefits to society. This in turn depends for example on the number of people within jurisdictions that benefit from the policy and have preferences for it.

In terms of *distribution of income*, a typical assumption within public finance is that the existence of equalization schemes in multiple fiscal constitutions indicates some societal preferences for equity (Boadway and Shah, 2009, chapter 1). The reduction of horizontal fiscal imbalances is seen a means to ensure a comparable provision of public goods and services among all citizens (Boadway and Shah, 2009, p. 233). As we have seen in the theoretical analysis, if the initial provision of the policy in question is larger where it is cheaper and the transfers benefits poorer states the most, a performance-oriented fiscal transfer would equalize available budgets among municipalities. But there are further implications to consider. If under municipal competition one municipality loses – even if it is the the one with highest budgets – this would be Pareto inferior to status quo if there was no compensation. But it might be Pareto superior. It depends on the relation of spill-over benefits which are provided by the provision in the neighbouring muncipality and decreased municipal budgets. When the worse-off municipality would benefit more through the spillover benefits than it would lose through a smaller own budget the situation would be Pareto superior. It would also pass an *actual* Kaldor-Hicks compensation test.[20] Note, however, that a corresponding assessment would have to quantify benefits obtained from an increased provision and compare those to lost income.

The third dimension, *systemic resilience*, refers to the contribution of performance-oriented fiscal transfer to the ability of a socio-ecological system to absorb shocks and return to a (quasi) equlibrium. In the literature it most often refers to economic stability (Zimmermann, Henke, and Broer, 2011) but it may as well be understood as ecological stability and ecosystem resilience. The latter has a variety of potential indicators depending on the type of disturbance and stabilizing elements. Nature-based solutions to

[20] A Kaldor-Hicks efficiency assumes that if the gain for one (market) participant sufficies for a hypothetical compensation for another participant that looses against status quo, this situation can be judged better than the one before (Hicks, 1939). With *actual* test I mean that there is not just an hypothetical compensation in this situation

counter-act ecological shocks could be seen as one possible way to increase ecosystem resilience but it may also refer to sustainable management practices to prevent overuse of natural systems. If a performance-oriented fiscal transfer would help to increase systemic resilience this could be understood as a contribution to a normative goal of an adaptive capacity of the overall socio-ecological system which reduces risks of large crises. As with the other dimensions, indicators will have to be employed to make systemic resilience operational.

For this dissertation, I will employ a set of criteria that are related to these dimensions of normative evaluation. The first is the *amount of protected area*, which can be considered a mixture of economic efficiency and systemic resilience since it is about net gains and ecological conservation. Protected areas can be understood as a public good with spillover benefits (ten Brink et al., 2013). Furthermore, the lack of progress towards (inter-)national strategic policy goals to halt biodiversity loss and enhance protected areas serves as a clear indication for a societal demand for more conservation and thus a current underprovision.[21] I will therefore assume that increasing the amount of protected area yields net gains. Additionally, protected areas are seen as one central measure to increase ecosystem resilience (Cumming et al., 2015; Newton, 2011). While data on spatially differentiated costs and preferences for conservation are difficult to obtain and will therefore not be part of the empirical analysis, the supposition is that a general increase in protected area enhances the supply of an underprovided public good which improves ecological resilience. The second, *distribution of income*, will be employed as a criterion for the analysis of simulated outcomes of policy proposals, i.e. chapter 6. Protected areas incur costs to the local level in terms of opportunity costs and mangement expenditures which constitute a fiscal need. If these are taken into account through EFT it may cover the fiscal needs but it may also have further distributional outcomes. Since fiscal needs can often only be assessed through proxies, I will basically assume that if EFT have an equalizing effect, this will be in line with the societal preference for fiscal balance and reduced inequality between municipalities. In other words, if EFT benefit poorer municipalities or admistrations the most, this can be judged beneficial given the existence of societal preferences for inequality reduction.

[21] As an example, the CBD's Aichi Biodiversity target number 11 is about safeguarding at least 17 per cent of terrestrial ecosystems, and 10 per cent of coastal and marine areas by 2020 (Convention on Biological Diversity, 2010).

Part III

Empirical Analyses

Municipal Responses to Ecological Fiscal Transfers in Brazil

A Microeconometric Panel Data Approach

This article has been published as

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Abstract: Ecological fiscal transfers in Brazil, the so-called ICMS-Ecológico or ICMS-E, redistribute part of the state-level value-added tax revenues on the basis of ecological indicators to local governments. We analyze whether the introduction of this economic instrument in a state offers incentives to municipal responses in terms of further protected area (PA) designation. We provide a microeconomic model for the functioning of ICMS-E and test the derived hypothesis empirically. Employing an econometric analysis on panel data for two decades we estimate the correlation of the introduction of ICMS-E in Brazilian states with PA coverage. We find that the introduction of ICMS-E correlates with a higher average PA share. While the introduction of ICMS-E schemes may be a compensation for a high share of federal and state PA, we also find an incentive effect for municipalities to designate additional PA.

Keywords: conservation incentives, ecological fiscal transfers, economic instruments, fiscal federalism, ICMS-Ecológico, policy evaluation

JEL codes: C23, H43, H77, Q28, Q57

3.1 Introduction

Against the backdrop of biodiversity and ecosystem service loss, suitable and effective policy instruments that could help to halt this trend are of great interest to meet the Aichi biodiversity targets of the Convention on Biological Diversity's Strategic Plan to 2020 (Convention on Biological Diversity, 2010). Ecological Fiscal Transfers (EFT) could be one such instrument. Intergovernmental fiscal transfers redistribute tax revenue from higher to lower levels of government, based on a number of different indicators such as population or area of the relevant jurisdiction. EFT redistribute a share of these public revenues according to nature conservation or other environmental indicators. Several authors see EFT as an instrument that could potentially incentivize greater nature conservation (Grieg-Gran, 2000; Loureiro, 2002; May et al., 2002; Ring, 2008c; Young, 2005). Since the pioneering implementation in the state of Paraná in 1991, a number of Brazilian states have adopted EFT from the state to the municipal level in the so called "Ecological Value-Added Tax" (ICMS-Ecológico, in short ICMS-E). To date 17 Brazilian states have implemented EFT schemes of which 16 have included explicit indicators relating to protected areas (PA) in the criteria for tax revenue distribution. This setting provides an opportunity to analyze the effectiveness of the instrument with regard to its economic incentive effect for designating additional PA.

The benefits of designated PA are mainly public in nature. However, the designation of further PA incurs opportunity costs. These are mainly costs to private actors such as land-use restrictions for agriculture, infrastructure, housing and industry. But they also lead to lower tax revenues for public jurisdictions and incur management costs for administrating bodies. This constitutes a problem of collective action and requires adequate institutions be put in place, defining who should be responsible for the required policies and who should bear their costs. The study of fiscal federalism analyzes how public functions and finance are and should best be distributed among different government levels in federal systems (Bird and Smart, 2002; Boadway and Shah, 2009; Musgrave, 1959; Oates, 1972, 2005). The principle of "fiscal equivalence" basically states that those who receive benefits of a policy should also pay for the related costs (Olson, 1969)[22]. In case of positive external effects beyond the boundaries of a jurisdiction that is paying for the provision of the relevant public good, this would require compensation payments. To address the issue, fiscal transfers are an adequate instrument to internalize spill-over effects (Bird and Smart, 2002; Boadway and Shah, 2009; Dahlby, 1996; Dur and Staal, 2008).

According to the Brazilian constitution the value-added tax (ICMS) is levied by states (*Constituição da República Federativa do Brasil de 1988*, Art. 155 II). A quarter of this relevant state revenue is allocated according to the derivation principle, it belongs to the municipalities that generated it (*Constituição da República Federativa do Brasil de 1988*, Art. 158 IV). Of this quarter, 75 per cent must be distributed proportionally to the contribution of each municipality to the value added of the state. The remaining 25 per cent (6.25 per cent of the total) is redistributed to municipalities according to criteria established under state law (e.g. population or agricultural production) (*Constituição da República Federativa do Brasil de 1988*). The ecological fiscal transfers (ICMS-E: Imposto sobre Circulação de Mercadorias e Serviços - Ecológico) introduce ecological criteria to redistribute this share, for instance considering registered PA on municipal territory.

Differing from state to state the share of the ecological indicator is up to 8 per cent of the municipal value-added tax revenue (2 per cent of total ICMS). The ICMS-E scheme was first implemented to reward municipalities for hosting (federal and state) PA, later on it was also thought to incentivize municipalities to designate additional municipal PA (Grieg-Gran, 2000; Loureiro, 2002; May et al., 2002; Ring, 2008c).

[22] This principle basically internalizes external effects of public policy. In the case of spill-over effects to other regions a more centralized government might be better suited to take account of the relevant public goods and services to avoid them being underprovided.

The scheme has several interesting attributes: i) it does not require any additional finance since it constitutes a change in the distribution of existing tax revenue – which is of particular interest due to the lack of conservation finance and overall budget constraints; ii) it partly decentralizes the decision of where to protect nature, taking into account local preferences and benefiting from local knowledge (Sauquet et al., 2014); iii) it is seen as an incentive for nature conservation and may provide a greater supply of an underprovided public good (Droste, 2013; Grieg-Gran, 2000; Loureiro, 2002; May et al., 2002; Ring, 2008c), iv) it potentially benefits low income municipalities that would not receive much (value-added) tax revenue in the absence of the instrument (Grieg-Gran, 2000), and v) the transaction costs for implementing such a scheme are considerably low since it represents only a rather marginal change in an existing fiscal transfer scheme (Ring, 2008c; Vogel, 1997).

EFT have recently gained quite some attention outside of Brazil. Portugal has established a municipal EFT scheme in 2007 (Santos et al., 2012). In France, there are compensation schemes for municipalities in core areas of national parks (Borie et al., 2014). In Queensland, Australia, a multi-criteria analysis has been used for the allocation of environmental funds via fiscal transfers (Hajkowicz, 2007). For Germany, Switzerland, Indonesia and India EFT schemes have been proposed to be introduced and the consequences simulated (Czybulka and Luttmann, 2005; Irawan, Tacconi, and Ring, 2014; Köllner, Schelske, and Seidl, 2002; Kumar and Managi, 2009; Mumbunan, 2011; Perner and Thöne, 2007; Ring, 2002, 2008c; Schröter-Schlaack et al., 2014). Farley et al. (2010) even suggest an adaptation to the global level. The studies for countries with implemented EFT schemes mainly focus on the institutional design of the instrument and provide limited empirical evidence of its effects on further PA designation. For Brazil, Sauquet et al. (2014) provide a first econometric analysis of the effects of the ICMS-E by analyzing strategic interaction among municipalities in the state of Paraná. The effectiveness of the socio-environmental ICMS in Pernambuco regarding social policies, namely education and health, has been studied by Da Silva Júnior and Sobral (2014) with a Markov Chain simulation.

This paper aims to contribute to the literature with an econometric approach that analyzes the effectiveness of the instrument on the basis of the introduction of the ICMS-E in 17 Brazilian states over the last two decades. The research question is whether the ICMS-E offers an incentive toward a local level response in terms of additional municipal PA designations? The econometric model estimates the correlation of introducing ICMS-E in Brazil with PA coverage in a panel data setting of all 27 Brazilian states from 1991-2009, controlling for socioeconomic and conservation policy variables. Hypothetically, municipalities are more inclined to increase their municipal-level PA or seek ways to attract federal or state designations of such PA if these become a source of income via EFT. The results will give insights about the functioning of the instrument, correlations of other variables with PA coverage, and provide lessons for the design of similar schemes.

The structure of the paper is as follows: Section 3.2 provides background information on the historical development and institutional details of ICMS-E schemes in Brazil. Section 3.3 briefly indicates the data source and gathering methods for the subsequent analysis. Section 3.4 presents a theoretical microeconomic model of the functioning of the instrument and describes the econometric model used to test our hypothesis generated from theory. Section 3.5 gives the result of the econometric analysis. Section 3.6 is about the discussion of our findings, their limitations and relevance, followed by our conclusion in section 3.7.

3.2 Background

The first ICMS-E scheme was introduced in Paraná, after a number of municipalities with PA for biodiversity conservation or watershed protection areas on their territory exerted pressure on the state government in 1990 (Grieg-Gran, 2000). ICMS revenue was largely distributed among the municipalities that generated it, while opportunity costs of PA were not taken into account. Municipalities with PA faced restrictions on land use and these were perceived as constraints in terms of both development and tax revenue generation. The mayors of the affected municipalities hence argued that complying with such land-use restrictions was difficult and demanded compensation (Grieg-Gran, 2000). In response, the first EFT scheme, with a 5 per cent share of the valued-added tax revenue accounting for the existence of PA for biodiversity conservation and watershed protection (2.5 per cent each), was implemented in late 1991 by the front-runner Paraná. The rationale for the first scheme was basically compensation for opportunity costs but it was soon thought of as an instrument that could also incentivize nature conservation (Grieg-Gran, 2000; May et al., 2002). After Paraná, São Paulo was the next state to introduce an EFT scheme in 1993 (with a relatively low ecological share of 0.5 per cent). Step by step other states followed and implemented similar EFT schemes, experimenting with different design options.

In some states the ICMS schemes incorporating environmental indicators are called socio-environmental ICMS (i.e. in Pernambuco and Ceará), and in the latter case it only refers to solid waste management. In Minas Gerais the law that includes the ICMS-E is officially called the 'Robin Hood Law', because it is designed to transfer tax revenues to poor regions and takes into account several social and environmental criteria; it was originally enacted in 1997 (Fundação João Pinheiro, 2014). The most commonly used method for determining the amount of EFT to be distributed to local governments largely builds on the pioneering example of Paraná (see Equations 3.1–3.3). An environmental index EI_i

$$\mathrm{EI}_i = \frac{\mathrm{MCF}_i}{\mathrm{SCF}} \tag{3.1}$$

is calculated as a ratio of municipality i's protected area (PA) portion of total municipal area (M), the municipal conservation factor

$$\mathrm{MCF}_i = \frac{\mathrm{PA}_i}{\mathrm{M}_i} \tag{3.2}$$

over the sum of all n municipalities' ratios, the state conservation factor

$$\mathrm{SCF} = \sum_{i=1}^{n} \mathrm{MCF}_i \tag{3.3}$$

while weighting different PA_i categories according to their contribution to conservation goals (cf. Loureiro, 2002; Loureiro, Pinto, and Motta, 2008; Ring, 2008c; Sauquet, Marchand, and Féres, 2014). As can be seen in Table 3.1, the institutional design of the ICMS-E schemes varies among states (cf. The Nature Conservancy, 2014).

Very important institutional features for the functioning of ICMS-E schemes are the specifics of the Brazilian conservation law and corresponding competencies of different government levels (cf. May et al., 2002). The National System of Protected Areas (SNUC) recognizes 12 different types of PA category, which roughly correspond to the PA category classification of the International Union for Conservation of Nature (IUCN). They are furthermore differentiated within two main groups: strictly protected areas (proteção integral), which in general do not allow private land ownership and land use, and less restrictive ones (uso sustentável), which permit sustainable land use and private property. While the first category is essential to protect endangered species and ecosystems and provides services only supplied by healthy and intact ecosystems, the latter category potentially increases the sustainability of, e.g., agricultural practices. A particular characteristic of the Brazilian system is that all three government levels have legislative powers to designate all 12 PA types on their own. Even the stricter categories such as 'parks' can be designated not just by the federal government but also by state and municipal governments. Furthermore, the SNUC devises rules for the voluntary designation of publicly recognized and legally binding PA on private land.

In 2014, a total of about 18 per cent of terrestrial area in Brazil was included under conservation statutes, which comprises about 8.8 per cent national PA, 8.9 per cent state PA and 0.3 per cent municipal PA (Ministério do Meio Ambiente, 2014a, see also next section for data collection methods). Although the latter seems insignificant, the huge spatial extent of Brazil makes 0.3 per cent equivalent to the area of Belgium. This is to say, compared with national and state conservation efforts, municipal conservation activities seem comparatively small. They may, however, engage local actors and their knowledge and thus protect viable spots for local ecosystem functioning and biodiversity protection (Grieg-Gran, 2000). Municipal PA are therefore not the most important ones from a large-scale contiguous conservation perspective, but they constitute an important and decentralized complement to the national and state governed PA systems.

Regarding the calculation of the ICMS-E, all different PA categories enter the computation of ecological fiscal transfers, but they may have different weights in states with different ICMS-E designs. Table 3.1 summarizes the shares and indicators of the different ICMS-E schemes in place.

TABLE 3.1: Introduction time and design of ICMS-E schemes in Brazilian states.

Brazilian states	*Year of first legislation*	*Year of legal enactment*	*Share of value-added tax for conservation efforts*	*Ecological indicators*
Acre (AC)	2004	2010	1% (2010), 2% (2011), 3% (2012), 4% (2013), 5% (from 2014)	PA (areas recognized in the national PA system and/or state system)
Alagoas (AL)	–	–	–	–
Amapá (AP)	1996	1998	1.4%	PA
Amazonas (AM)	–	–	–	–
Bahia (BA)	–	–	–	–
Ceará (CE)	2007	2008	0% (only solid waste management is considered)	waste manangement
Espírito Santo (ES)	–	–	–	–
Federal District of Brasília (DF)	–	–	–	–
Goiás (GO)	2011	2012	up to 5% in form of a composite indicator (1.25 in 2012, 2.5% in 2013, 3.75% in 2014, 5% in 2015	sustainable development plans (PA, waste management, environmental education, reduced deforestation, reduced forest fires, watershed protection etc.)
Maranhão (MA)	–	–	–	–
Mato Grosso (MT)	2000	2002	5%	PA and indigenous land
Mato Grosso do Sul (MS)	1994	2002	2% (2002), 3.5% (2003), 5% (2004) for various environmental criteria	PA, indigenous lands, waste management plans
Minas Gerais (MG)	1995	1997	PA 1 of 3 environmental criteria 0.5% (2010), 0.45% from 2011	PA per municipal area, conservation factor (PA category) and conservation quality factors
Pará (PA)	2012	2014	for all environmental criteria 2% (2012), 4% (2013), 6% (2014), 8% (from 2015)	PA extent, avoided deforestation, registered rural lands etc.
Paraíba (PB)	2011	not yet	5%	PA
Paraná (PR)	1991	1992	2.5% for PA for biodiversity conservation and 2.5% for PA for watershed	PA, PA category, and variation of conservation quality
Pernambuco (PE)	2000	2001	1% PA share per municipal area, their category and degree of conservation	
Piauí (PI)	2008	2009	overall environmental criteria are 1.5% in 2009; 3.5% in 2010; 5% from 2011 (PA 1 out of 9 environmental criteria)	Waste management, watershed protection, reducing deforestation, pollution control, PA, etc.
Rio de Janeiro (RJ)	2007	2009	1% (2009), 1.8% (2010), 2.5% from 2011	PA, water quality, waste management, plus an extra for designation of municipal PA
Rio Grande do Norte (RN)	–	–	–	–

Continued on next page

Brazilian states	*Year of first legislation*	*Year of legal enactment*	*Share of value-added tax for conservation efforts*	*Ecological indicators*
Rio Grande do Sul (RS)	1997	1998	7% (for a composite indicator)	Municipal area, 3 times PA, indigenous lands, inundated lands
Rondônia (RO)	1996	2003	5% Share of PA per municipal area, number of PA and past year total PA area	
Roraima (RR)	–	–	–	–
Santa Catarina (SC)	–	–	–	–
São Paulo (SP)	1993	1994	0.5% only accounting for state PA	PA and PA category
Sergipe (SE)	–	–	–	–
Tocantins (TO)	2002	2007	3.5%	PA and indigenous land (+ another 3.5 for watershed protections, waste management, etc.)

Source: authors' elaboration based on Nature Conservancy (2014) and legislative acts.

3.3 Data Collection

This study builds on the analysis of legal documents regarding the introduction of state-level ICMS-E schemes and is based on data for PA coverage, socio-economic data of the share of value added by different sectors, population density and per capita GDP. The Nature Conservancy website (2014) on the ICMS-E schemes provides background information and links to the legal documents in which the schemes are specified. We have collected data on both the original state law that basically prepares the legal grounds and on the implementing decrees that actually enact the schemes. The national cadaster of conservation units (CNUC) of the Brazilian Ministry of the Environment (Ministério do Meio Ambiente, 2014a) provides data on PA recognized within the national system of conservation units (SNUC) with respect to time of enactment, area and related legal acts. We have furthermore consulted the state environmental secretariats' websites for complementing the national cadaster, because the latter relies on input from the governing bodies and apparently is not entirely complete, i.e. with regard to municipal PA. We only complemented the national cadaster data when a) the category of additional PA complies with the national system of conservation areas, b) there was data on the area, and c) legal acts were indicated. We then used the i3Geo software of the Brazilian Government (Ministério do Meio Ambiente, 2014b) to calculate the terrestrial PA size. We excluded marine PA, since they are not included in any ICMS-E scheme and are furthermore large in their area. Where we identified a spatial overlap in the available geo-referenced data we computed the overlap free PA share with the following hierarchy: i) in case there is both strict protection and sustainable use area we only accounted for the first; ii) in case there are two PA of the same category from different government levels we only accounted for the highest government level PA. We thereby compiled a data set of the development of PA at different government levels and their share of state territory by year among Brazilian states. Note that in Brazil, the local governmental level can also designate municipal-level PA of all categories. The governmental Institute for Applied Economic Research (IPEA) provides data on the value added by agriculture and industry, on estimated population density and on GDP per capita of the Brazilian states for 1991-2009 (IPEA, 2014). All this data was gathered in a panel data set for the Brazilian states.[23]

3.4 Theoretical and Econometric Models

3.4.1 Theoretical Model

The theoretical model of the effects of fiscal transfers is based on Boadway and Shah (2009, chapter 9) and represents a simplistic microeconomic model of a government body's spending behavior receiving a fiscal transfer (see Figure 3.1). The model substantiates the derivation of hypotheses that are tested in the empirical part (section 3.4.2 and 3.5).

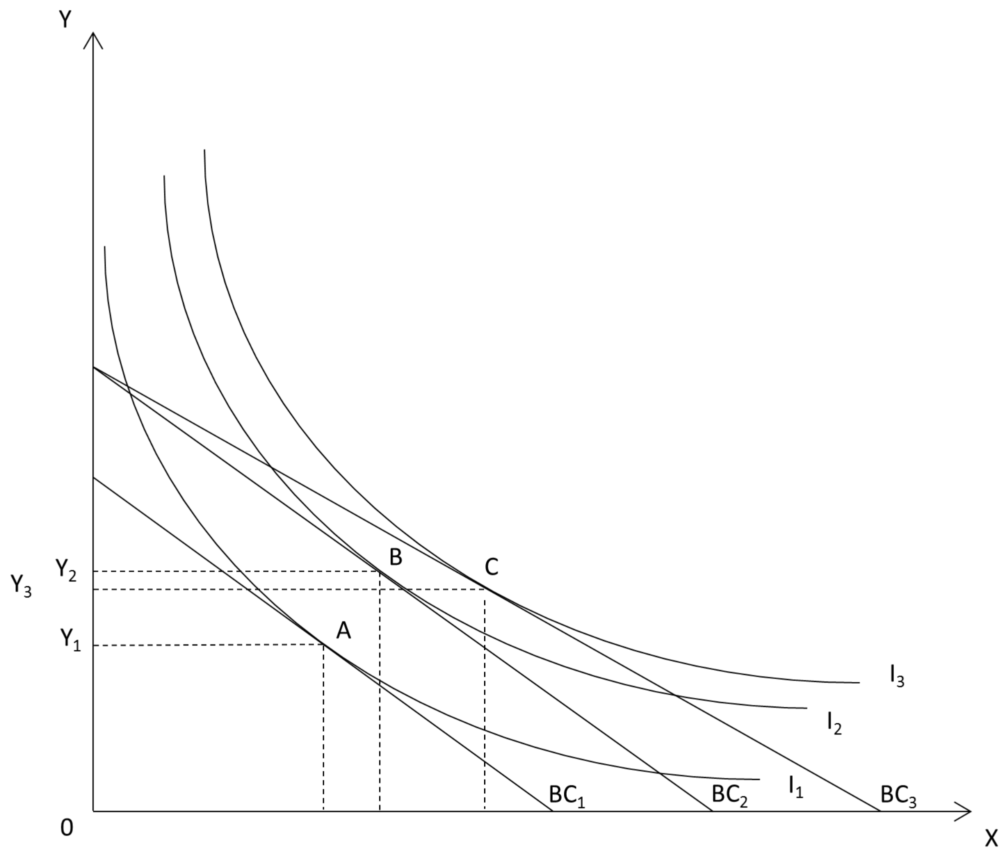

FIGURE 3.1: Government spending behavior after introduction of fiscal transfers. *Source*: authors' work adapted from Boadway and Shah (2009, chapter 9).

[23] The dataset can be downloaded at: `https://github.com/NilsDroste/EFT-BR`

The ICMS-E in Brazil is a general budget support based on the share of PA on the municipal territory. When a government body receives such an unconditional general purpose transfer there are no obligations for a particular spending behavior, i.e. they are not ear-marked for specific purposes. Let us consider this in a two-good model world. Let Y denote a composite non-nature conservation public good or service and X a nature conservation public good or service that both can be supplied by a government body by some sort of public expenditure. Let there be a budget constraint BC representing the amount that can be spent based on relative prices and a utility indifference curve I based on the utility gained by the satisfaction of preferences and the marginal rate of substitution among these two goods. A (boundedly) rational government would maximize its utility where the marginal rate of substitution equals the budget constraint (see point A in Figure 3.1). A general purpose transfer to this government shifts the budget constraint outwards, e.g. from $BC1$ to $BC2$. The government's budget increases with the transfer as does spending on both goods. Now, the ICMS-E is a general purpose transfer based on a particular public service, namely the existence of PA on the territory of the municipality. This represents a price change in terms of relative prices since every unit of PA will be rewarded and a PA designation therefore has lower opportunity costs than without the instrument. In our model world this can be seen as the price change ($BC2$ to $BC3$). The spending effect of such a shift will reduce spending on Y ($Y2$ to $Y3$) and increase spending on X. Comparing the initial state without the fiscal transfer (point A) and the state with fiscal transfers (point C) spending on both goods increases. The effect of the fiscal transfers can be decomposed into two partial effects, namely the outward shift of the budget constraint and the price change. The empirical outcomes inter alia depend on the real structure in preferences, relative prices, marginal rates of substitution, and extent of rationality applied. Real world outcomes may be more complex than this simplified model suggests.[24]

Nevertheless, the hypothesis that we derive from this model is that if there is an ICMS-E scheme in place municipal governments receiving the EFT will increase spending on both non-nature conservation and nature conservation. We suppose that such an increased spending on nature conservation should to some extent be reflected in an increased share of PA on the municipal territory, i.e. because they constitute a source of additional income. But EFT might also decrease the municipal resistance to accepting a state or federal PA on municipal territory due to the 'price effect' of lower opportunity costs for hosting PA. Therefore, we hypothesize that in states and years where there is an ICMS-E scheme in place a higher (municipal) share of PA should be observed.

[24] We recognize the possibility of a strategic interaction: The probability of a municipality receiving extra income through conservation action is lowered by the entrance of other municipalities' conservation action. Nevertheless, the amount of money distributed through the ICMS-E system also depends on the performance of the economic system since ICMS-E is a share of the total value-added tax revenue. In case of economic growth more money will be distributed according to PA shares. Strategic conservation interactions have partly been addressed by Sauquet et al. (2014), but to our knowledge an analysis of thresholds at which a municipality does not get any further transfer when designating an additional PA has not yet been conducted and remains for future research.

3.4.2 Econometric model

The econometric model estimates the correlation of ICMS-E schemes with nature conservation area share with random effects regressions controlling for other conservation instruments, biomes, and socio-economic variables for land-use pressure such as the share of value added by agriculture and industry, population density, and GDP per capita. The general structure of the regression is as outlined in equation 3.4.

$$\begin{aligned}\ln(\text{PA})_{it} = \beta_o + \beta_1\text{icms\_e}_{it} + \beta_2\ln(\text{agr})_i t + \beta_3\ln(\text{ind})_{it}\\ +\beta_4\ln(\text{pop})_{it} + \beta_5\ln(\text{inc})_{it} + \beta_6\text{arpa}_{it}\\ +\beta_7\ln(\text{oPA})_{it} + \beta_8\text{biome}_{ji} + \beta_9\text{year}\\ +\beta_{10}\text{int}_{it} + \mu_i + \epsilon_{it}\end{aligned} \tag{3.4}$$

where $i = 1, ..., n$ indexes the Brazilian state, $t = 1, ...T$ indexes years, and $j = 1, ..., k$ indexes the biomes. PA_{it} is the share of PA in year t of total territory of state i in per cent which will either be PA total (PAtot), federal (PAfed), state (PAsta) or municipal (PAmun) PA share.[25] The policy variable icms_e is a dummy with the value of 1 in case of an existing ICMS-E scheme in state i in year t and 0 if otherwise. The socio-economic controls for states i and years t, agriculture agr and industry ind are the per cent shares of total value added by these two sectors (at constant prices of year 2000)[26] pop is population density in inhabitants per km^2, and inc is GDP per capita at constant prices of year 2002.[27]; arpa is a dummy variable for the ARPA policy,[28] which supports PA designations in the Amazon with a value of 1 in case of implementation in state i in year t and 0 if otherwise. Other PA oPA is only included when regressing federal, state or municipal PA on the explanatory variables and consists of a vector of the other two PA share variables, e.g. federal level PA share PAfed, and state level PA share sta in case of a regression of municipal PA share PAmun on the controls. The vector of dummy variables biome indicates a minimum 5 per cent share of any of the 6 different biomes of Brazil on state i's area (IBGE and MMA), 2004): Amazon (ama), Cerrado (cer), Caatinga (caa), Atlantic Forest (mat), Pantanal (pan), and Pampa (pam) – which can be overlapping in case a state has different biomes.[29] The year variable is included

[25] In case of state and municipal PA there are reasonable zeros in the data. We added a constant c of half the minimum observed value for each state and municipal data to allow for log transformation.

[26] We originally included both the industry and the service sector in the model. However, the logarithms of the service sector and the industry sector value-added variables are strongly correlated with a coefficient of 0.81. In order to avoid multicollinearity we only included the industry variable in the regression.

[27] The data was given in year 2010 prices and has been recalculated with consumer price indices (IBGE, 2015).

[28] ARPA means the Amazon Region Protected Areas Programme, which collaborates with state and local governments in creating new PA in the Amazon biome, and has come into force in 2002.

[29] The value of the biome dummies is set according to the following categorization (see table 3.1 for abbreviation of the states): RO=ama, AC=ama, AM=ama, RR=ama, PA=ama, AP=ama, TO=ama&cer, MA=ama&cer, PI=cer&caa, CE=caa, RN=caa&mat, PB=caa&mat, PE=caa&mat,

in individual random effects regressions to detrend the development of PA designations. Variable int stands for a vector of interaction variables of the above-mentioned icms_e and the socio-economic control variables. The error terms are: μ_i, individual error term and $\epsilon_i t$, idiosyncratic error term – making it a one-way (individual specific) effects regression (Croissant and Millo, 2008). The panel is balanced with $n = 27$ for the 26 Brazilian states and the Federal District of Brasilia, $T = 19$ for the years $1991 - 2009$, and $N = 513$ observations in total. Hypothetically, states may have an incentive to enhance PA coverage if PA become a source of income by the EFT scheme – hence, we should be able to observe a correlation of ICMS-E with PA coverage (H_1). The null-hypothesis (H_0) is that an increase in budget due to ICMS-E does not correlate with PA coverage – which could mean that the additional ICMS-E income has very likely been spent on different non-nature conservation public services.

The regressions are computed with the **plm** package (Croissant and Millo, 2008) in **R** (R Development Core Team, 2013) with random effect regressions to control for unobserved individual heterogeneity such as state preferences for nature conservation (Wooldridge, 2010, chapters 10, 11). Standard errors are computed with covariance matrix estimators robust to heteroskedasticity, serial and spatial (cross-sectional) correlation, with a maximum lag window of $m(T) = 2$ (Driscoll and Kraay, 1998; Millo, 2016)[30] and heteroskedasticity consistent covariance estimation type $HC3$, which gives less weight to influential observations (Long and Ervin, 2000; Zeileis, 2004).[31]

3.5 Results

We estimate the effect of ICMS-E schemes and several political, socio-economic and geographic indicators on the average PA share of 26 Brazilian states plus the Federal District for the years $1991 - 2009$. Summary statistics are provided in the appendix. For both regressions of total PA share and municipal level PA share (Tables 3.2 and 3.3), we start with a simple model including the socio-economic control variables (Model 1) in

AL=caa&mat, SE=caa&mat, BA=cer,caa&mat, MG=cer&mat, ES=mat, RJ=mat, SP=cer&mat, PR=mat, SC=mat, RS=mat&pam, MS=cer,mat&pan, MT=ama,cer&pan, GO=cer, and DF=cer.

[30] Given a rejection of the null hypothesis of a unit root at the 5 per cent significance level with a cross-sectionally augmented Im–Pesaran–Shin unit-root test for single time series of panel data (Im, Pesaran, and Shin, 2003; Pesaran, 2007), the corresponding order of integration for each of the logarithmized non-stationary time series is given in parenthesis: PAtot (1), PAfed (1), PAsta (> 2), PAmun (2), agr (1), ind (1), pop (1), inc (1).In contrast, a panel covariate augmented Dickey–Fuller test on the entire panel (Demetrescu, Hassler, and Tarcolea, 2006; Kleiber and Lupi, 2011) rejects the null at 1 per cent significance levels. However, since we are rather interested in long-term effects, a first-differencing approach – which would generally be a way to go for non-stationary time series – does not seem appropriate. As has been shown (Pesaran and Smith, 1995; Phillips and Moon, 1999) long-run relationships can consistently be estimated with (quasi-)demeaned data through fixed or random effects and detrended data, given cross-sectional independence. Therefore, in our case of cross-sectional dependence, a robust estimation is a must. The Driscoll and Kraay (1998) covariance matrix estimation employed accounts for auto- and cross-sectional correlations and even for spatial dependence. We therefore consider it a suitable approach for handling the various sources of dependencies across estimates.

[31] Both the R code and the data required to reproduce the results presented in this paper can be found at `https://github.com/NilsDroste/EFT-BR`.

which we add-in further control variables for the biomes (Model 2), the year variable (Model 3), and interaction terms (Model 4).

We find a significant and positive correlation of icms_e, significant and negative correlation of ln(agr) and a significant positive correlation of ln(inc) with the logarithm of PA share ln(PAtot) (see table 3.2). We furthermore find structural differences of how much nature is conserved among the biomes of Brazil. As soon as the biome dummies are included the explanatory power of the model increases (Model 2). The numeric year variable is positively and significantly correlated and its interaction term with icms_e is negatively and significantly correlated (Model 3). Furthermore, ln(ind) has a positive and significant correlation when year and its interaction term are included (Models 3 and 4). An inclusion of further interaction terms of icms_e with socio-economic controls does not yield any further significant correlation or increase in explanatory power (Model 4).

TABLE 3.2: Overall protected area share and ICMS-E

	Dependent variable: **ln of protected area share in percent of total area**							
	(1)		(2)		(3)		(4)	
icms_e	0.549**	(0.223)	0.512***	(0.105)	1.146***	(0.285)	2.490	(1.525)
ln(agr)	−0.372***	(0.113)	−0.294***	(0.081)	−0.202***	(0.072)	−0.200***	(0.077)
ln(ind)	−0.087	(0.119)	−0.018	(0.127)	0.217*	(0.114)	0.235**	(0.103)
ln(pop)	0.199***	(0.052)	0.643***	(0.154)	−0.079	(0.122)	−0.058	(0.144)
ln(inc)	2.164***	(0.532)	2.513***	(0.240)	0.891*	(0.486)	0.884**	(0.445)
arpa	0.220	(0.217)	−0.033	(0.129)	−0.154	(0.161)	−0.202	(0.222)
ama			3.251***	(0.507)	1.276*	(0.731)	1.229*	(0.745)
cer			0.116	(0.359)	0.448***	(0.120)	0.480*	(0.258)
caa			1.864***	(0.421)	−0.104	(0.471)	−0.210	(0.615)
mat			−0.859**	(0.372)	−0.685***	(0.181)	−0.680***	(0.194)
pan			−1.431**	(0.595)	−1.929	(1.976)	−1.976	(2.329)
pam			−1.035	(0.771)	−0.713	(0.531)	−0.600	(0.400)
year					0.090***	(0.011)	0.092***	(0.011)
icms_e*year					−0.063***	(0.011)	−0.055***	(0.017)
icms_e*ln(agr)							0.142	(0.129)
icms_e*ln(ind)							−0.143	(0.444)
icms_e*ln(pop)							0.065	(0.124)
icms_e*ln(inc)							−0.865	(0.614)
Intercept	−1.461	(1.795)	−4.660***	(0.885)	−0.845	(1.374)	−0.909	(1.187)
Adjusted R^2	0.372		0.451		0.569		0.574	
F Statistic	51.469*** (df = 6; 506)		36.038*** (df = 12; 500)		49.284*** (df = 14; 498)		39.396*** (df = 18; 494)	
Effects	individual re		individual re		individual re		individual re	

The panel data sample is balanced with $n = 27, T = 19, N = 513$. Robust standard errors are reported in parentheses. Individual coefficients are indicated with $^{*}10\%$; $^{**}5\%$; $^{***}1\%$ significance levels. Models use random effects (re) specifications

Regarding the effect of the ICMS-E on municipal-level PA designation, we find a significant and positive correlation of icms_e, a positive and significant correlation of ln(agr) and ln(pop) and a negative significant correlation of ln(ind) with the municipal PA share (Models 1 and 2 in table 3.3). We furthermore find a positive significant correlation of the natural logarithm of federal level PA share ln(PAfed). Including the time trend does not change much (Model 3). Once the interactions of icms_e with other socio-economic controls are included the picture becomes more complex: e.g., both the time trend and its interaction term show a significant and positive correlation; the interactions with ln(agr), ln(pop) and ln(PAfed) are significant and negative (Model 4).

TABLE 3.3: Municipal level protected area share and ICMS-E

	Dependent variable: **ln of municipal protected area share in percent of total area**							
	(1)		(2)		(3)		(4)	
icms_e	1.332***	(0.189)	1.275***	(0.177)	1.095**	(0.486)	4.834	(5.637)
ln(agr)	0.846***	(0.232)	0.924***	(0.229)	1.014***	(0.257)	0.856***	(0.186)
ln(ind)	−1.326***	(0.372)	−1.524***	(0.405)	−1.324***	(0.501)	−1.188***	(0.369)
ln(pop)	2.000***	(0.498)	2.465***	(0.615)	1.955***	(0.553)	1.846***	(0.670)
ln(inc)	2.536***	(0.706)	2.115***	(0.772)	1.205*	(0.717)	0.092	(0.531)
ln(PAfed)	0.648***	(0.193)	0.662***	(0.197)	0.596***	(0.195)	0.614***	(0.151)
ln(PAsta)	0.116	(0.101)	0.125	(0.108)	0.095	(0.134)	0.079	(0.129)
arpa	0.512	(0.334)	0.388	(0.372)	0.368	(0.387)	1.243**	(0.574)
ama			−4.877*	(2.486)	−6.177***	(1.695)	−5.951***	(1.094)
cer			−5.344***	(1.164)	−5.074***	(1.310)	−4.713***	(0.542)
caa			−0.763	(1.944)	−2.089	(1.556)	−3.274***	(1.174)
mat			−3.697*	(2.145)	−3.664*	(2.163)	−2.274	(1.411)
pan			4.956	(6.545)	4.314	(7.358)	2.778	(2.586)
pam			−1.476	(2.562)	−1.398	(2.727)	−3.268	(2.578)
year					0.054	(0.046)	0.061*	(0.036)
icms_e*year					0.008	(0.030)	0.089***	(0.025)
icms_e*ln(agr)							−0.663***	(0.240)
icms_e*ln(ind)							0.162	(1.081)
icms_e*ln(pop)							−1.006**	(0.461)
icms_e*ln(inc)							1.548	(1.670)
icms_e*ln(PAfed)							−2.437***	(0.567)
icms_e*ln(PAsta)							−0.356	(0.307)
Intercept	−14.919***	(2.521)	−9.591**	(4.586)	−7.292**	(3.541)	−5.853**	(2.441)
Adjusted R^2	0.349		0.364		0.365		0.427	
F Statistic	35.274*** (df = 8; 504)		21.963*** (df = 14; 498)		19.409*** (df = 16; 496)		18.369*** (df = 22; 490)	
Effects	individual re		individual re		individual re		individual re	

The panel data sample is balanced with $n = 27, T = 19, N = 513$. Robust standard errors are reported in parentheses. Individual coefficients are indicated with *10%; **5%; ***1% significance levels. Models use random effects (re) specifications

3.6 Discussion

First of all, there are some methodological remarks to consider. Generally, the panel data setting with individual effects would allow for estimating causal effects, or the 'average treatment effect on the treated' with Brazilian states self-selecting into 'treatment' of implementing ICMS-E schemes compared to an average of states without such a scheme (see, e.g., Wooldridge, 2010, Chapter 18 for a discussion of causal effects estimations). This analysis of the ICMS-E effects, however, is among the very first of its kind and there are no reference models neither on causal factors for protected area designation nor on the causal effects of policies such as ICMS-E schemes on protected area coverage (except to some extent Sauquet et al. 2014 who focus on spatial interaction). We therefore tend to be cautious on the issue and speak of an observed correlation rather than a causal effect.

With regard to the coefficients and their magnitude we also tend to be cautious and will not elaborate too much on the strength of the marginal effects[32] of introducing ICMS-E schemes, because we know that there is data missing for municipal PA and this may bias estimations (see below for further limitations). For the purpose of illustrating the apparent functioning of the ICMS-E schemes we mainly focus on the direction of correlations, that is to say the signs and significance. Figure 3.2 furthermore provides an overview about interactions which we discuss for some examples found in the data in more detail. Consider the ICMS-E dummy variable D in Figure 3.2. If there is an interaction with ICMS-E this means that the intercept will be $\beta_0 + \beta_1$ and the slope $\beta_2 + \beta_4$, and for the cases where there is no ICMS-E it will be β_0 and β_2 , respectively. In the following we discuss both the overall PA share and municipal PA share regressions as well as some overarching issues such as reverse causality and inferences.

Overall PA Share Regressions

We find a positive and significant correlation of the existence of an ICMS-E scheme with overall PA share among Brazilian states for the years $1991 - 2009$. This means on average there is a higher PA ratio in states and years with an ICMS-E scheme in place. Furthermore, we find one significant correlation in all regressions: a positive correlation of PA coverage with GDP per capita. This means that, on average, where and when there is a higher per capita income more PA are observed. A potential explanation may be that nature conservation is not the first thing to think about when there is no or little income available. Once basic needs are covered a healthy environment becomes more important.

[32] When both dependent and independent variables are log transformed, the coefficients can be interpreted as a percentage change, say a 1% change in agr corresponds to a $[(1.01)^{\beta 1} - 1] \times 100$ percentage change in PAtot holding everything else constant, for PAtot $= \beta_0 + \beta_1$agr $+ u_i$. Note, however, that coefficients of binary variables have to be interpreted as $100[\exp(\hat{c} - 0.5\hat{v}(\hat{c}) - 1]$; where $\hat{v}(\hat{c})$ is the estimated variance of $\hat{c}$ or the square of the standard error (Giles, 2011; Kennedy, 1981).

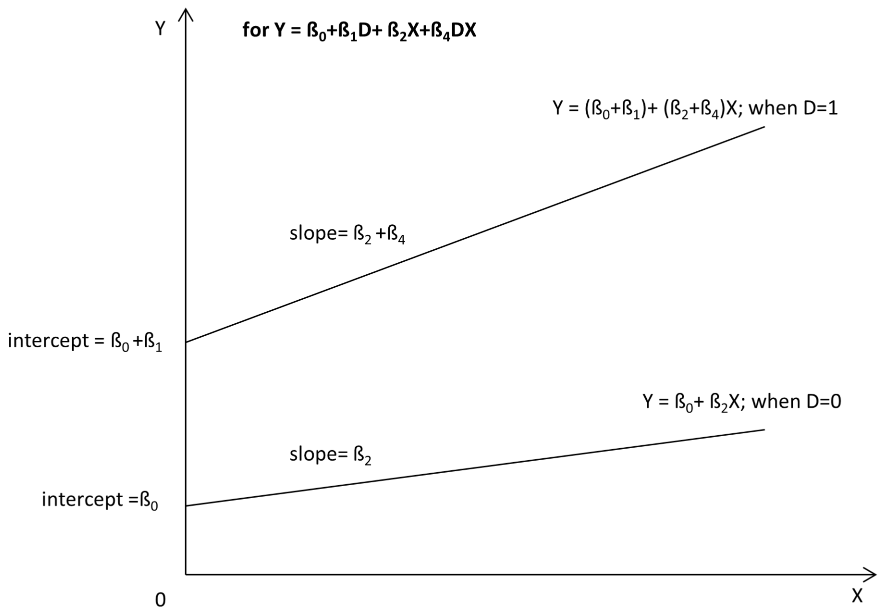

FIGURE 3.2: The interaction of the ICMS-E dummy variable D with a continuous variable *Source*: authors' work adapted from Brambor (2005).

Municipal PA Share Regressions

On the municipal level the regressions reveal complex patterns. We find a positive and significant correlation of the share of value added by agriculture, population density and GDP per capita, and a negative significant correlation of the share of value added by industry with the municipal PA share. On average, the share of value added by industry constitutes the highest opportunity costs for municipal PA designation. Furthermore, municipalities with high population density more often designate PA. When interaction terms with socio-economic controls are included the coefficients of the ICMS-E interaction with both agriculture and population density are significant and negative. Thus, where there is an ICMS-E scheme in place the direction of the correlation may change. Considering Figure 3.2 one can interpret the correlation of agriculture with municipal PA share ($\beta_{\text{ln(agr)}} = 0.856$) where there is an ICMS-E scheme ($\beta_{\text{icms\_e*ln}(agr)} = 0.663$) as still positive but with a lesser effect ($\beta_{\text{ln(agr)}}) + \beta_{\text{icms\_e*ln(agr)}} = 0.193$) (see Model 4 in Table 3.3). A similar pattern applies to the correlation of population density: the effect with an ICMS-E in place appears to be weaker but in the same direction ($\beta_{\text{ln(pop)}} + \beta_{\text{icms\_e*ln(pop)}} = 1.846 - 1.006 = 0.840$). Additionally, we find a positive significant correlation of federal level PA share with municipal level PA share and a significant negative interaction of federal level PA coverage with ICMS-E schemes. This means that, on average, the ICMS-E creates a crowding out effect of federal (and state) PA on municipal PA – which constitutes some sort of government level competition. This crowding out pattern may relate to the relative scarcity of available area and only becomes apparent once there is no longer abundant area available for conservation.

Overarching Issues

There are positive and significant correlations of ICMS-E schemes with both total and municipal PA coverage. However, this may relate to a reverse causal effect such that the introduction of ICMS-E is following the designation of a large share of PA instead of the ICMS-E providing an incentive to designate additional PA. Therefore, we included a time variable to account for trends in the PA data and an interaction of the ICMS-E dummy with the time variable. For total PA coverage the time variable is positively and significantly correlated and its interaction term with ICMS-E is negatively and significantly correlated (Table 3.2). This means that the average yearly increase in overall PA share is lower once an ICMS-E scheme is in place. Although this may also be due to increasing opportunity costs of additional PA designation, the reverse causal relation cannot be ruled out for the overall PA share. ICMS-E could be a consequence of an above average designation of federal and state level PA. In such a case, the additional budget may rather compensate for corresponding land-use restrictions and nevertheless change the mind-set of local governments, which commonly perceive PA as an obstacle to economic development. In fact, for municipal PA coverage the pattern is different. Both the time variable and its interaction term have positive and significant coefficients once other interactions are included. This means that the average yearly increase in municipal PA share increases once there is an ICMS-E scheme. This indicates an incentive effect for municipalities to designate additional PA when there is an EFT scheme in place. Altogether, this means that the hypothesis we derived from the simplistic microeconomic model presented earlier cannot be falsified. Although the ICMS-E schemes may be a consequence of a high overall share of PA, e.g. national and state level designations, on average we observe a local response to the creation of an EFT. Municipalities designate additional PA. This also means a decentralization effect regarding the location of further PA, because the additional ones are designated at municipal level. We would cautiously infer these to be a consequence of the fiscal incentive effect inherent in the ICMS-E scheme. By acknowledging the spill-over benefits of conservation and compensating for the local costs associated with conservation through a share of fiscal transfers, the public good provision of municipal PA is increased. Last but not least, we also have to comment on the limitations of our analytical approach. Although we used the most complete data available there may be missing data points, i.e. for municipal protected areas. We found indications that there are more municipal PA, but could not gather information on either their area or their year of enactment or corresponding legal acts and therefore refrained from including these. We consider this a potential source for biased estimates (see, e.g., the large increase in the ICMS-E dummy variable and its standard deviation in Model 4, Table 3.2). Although we have computed standard errors robust to spatial or cross-sectional and serial dependence (Driscoll and Kraay, 1998; Millo, 2016), which are quite conservative, this might still be a potential source of biased estimates (King and Roberts, 2015). A task for future research is to employ spatial estimations for panel data (cf. Millo, 2014) regarding the ICMS-E schemes in Brazil. We also did not include a continuous variable of ICMS-E schemes that accounts for the different institutional designs (see Table 3.1) and could give evidence of the strength of different ICMS percentages, since it is not in all cases clear what percentage is finally dedicated for the existence of PA. One particular aspect is worth mentioning, since the quantitative analysis conducted does not take into account the quality of the management of PA. Only in Paraná are quality criteria already included in the ICMS-E law (Loureiro, Pinto, and Motta, 2008). The fact that the ICMS-E apparently leads to an increase in PA does not necessarily mean that the current land-use practice is altered much – although we suspect that the designation of a PA helps nature conservation, the topic of management quality has not been touched upon by our analysis.

3.7 Conclusion

We analyzed the effect of introducing ecological fiscal transfers in Brazilian states for panel data covering the years $1991 - 2009$, following the pioneering introduction of ICMS-E in Paraná. Our research question was whether the ICMS-E creates an incentive to designate further PA. We presented a simplifying microeconomic model and tested the derived hypothesis econometrically controlling for unobserved individual, time, socio-economic variables, and other conservation policies. We find that the introduction of ICMS-E schemes on average corresponds, ceteris paribus, to higher total PA coverage. This could be a consequence of an ICMS-E introduction following a high PA share, that is to say a compensation for hosting other government level PA. On the municipal level, however, there are clear indications for local responses to the implementation of EFT: after an ICMS-E introduction additional municipal PA are designated. This signals a decentralizing effect for nature conservation. Both observations are very likely a consequence of the incentive effect inherent in ICMS-E schemes. We thereby have contributed to the literature with a first comprehensive econometric approach covering all Brazilian states and provide insights by providing a first estimation of the effects of introducing ICMS-E schemes on PA designation. The results of this study may thus advance the implementation of EFT schemes in other Brazilian states or other nations and are particularly relevant for countries in which an introduction might be expected (e.g. Germany, Poland) (Schröter-Schlaack et al., 2014). Especially since EFT schemes do not require any additional budget but constitute a (rather marginal) change in the allocation of tax revenue, they are relatively easy to implement. EFT schemes are thus of eminent relevance for conservation policies regarding a common shortage in public budgets at local levels and a shortfall of conservation budgets. Through incentivizing municipal PA designations EFT could help the implementation of (inter)national biodiversity targets such as the Aichi targets, the goals of the EU Biodiversity strategy, and national biodiversity strategies and action plans.

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Appendix. Descriptive Statistics

TABLE 3.4: Summary statistics

Statistic	N	Mean	SD	Min	Max
total protected area share of state territory in per cent (*PAtot*)	513	11.2	15.2	0.05	98.9
federal protected area share of state territory in per cent (*fed*)	513	6.9	12.9	0.03	92.9
state protected area share of state territory in per cent (*PAsta*)	513	4.0	4.9	0.0	22.8
municipal protected area share of state territory in per cent (*PAmun*)	513	0.2	1.1	0.0	7.5
ICMS-E dummy (*icms_e*)	513	0.2	0.4	0	1
share of valued added by agriculture in per cent (*agr*)	513	10.7	7.2	0.2	41.5
share of valued added by industry in per cent (*ser*)	513	27.7	11.5	3.6	66.1
share of valued added by service in per cent (*ind*)	513	61.5	12.5	31.7	96.0
population density cap/km² (*pop*)	513	26.3	31.5	0.9	174.2
GDP per capita, R$ in thousands (*inc*)	513	5.8	3.9	1.8	24.2

Source: authors' calculation based on Nature Conservancy (2014) and IPEA (2014). Monetary values in constant prices (2000 *R*$).

Decentralization Effects in Ecological Fiscal Transfers

The Case of Portugal

This article is based on a working paper, has been submitted to Environmental and Resource Economics and received a major revision decision. This is the revised version.

Droste, N., Becker, C., Ring, I., Santos, R. (2017) Decentralization effects in ecological fiscal transfers – the case of Portugal. *UFZ Discussion Paper 3/2017*.

Abstract: Portugal has a unitary system in which the central government transfers funds to lower government levels for their public functions. In 2007, Portugal introduced Ecological Fiscal Transfers (EFT), where municipalities receive transfers for hosting Protected Areas (PA). We study whether introducing EFT in Portugal incentivized municipalities to designate PA and has led to a decentralization of conservation decisions. We employ a Bayesian structural time series approach to estimate the effect of introducing EFT in comparison to a simulated counterfactual time series. Quantitative results show a significant increase in the ratio of municipal and national PA designations following Portugal's EFT introduction – which we infer to be a causal consequence. The analysis furthermore places emphasis on the importance of relevant municipal conservation competencies for the functioning of the instrument. Results have important implications for conservation policy-making in terms of allocating budgets and competencies in multi-level governments.

Keywords: Bayesian structural time series, ecological fiscal transfers, fiscal federalism, municipal competencies, nature conservation, Portugal

JEL codes: C32, H41, H72, Q57

4.1 Introduction

In the face of a rapid biodiversity loss (MEA, 2005) and the increasingly recognized importance of ecosystem services for human well-being (MEA, 2005; TEEB, 2010), the role of public conservation becomes by no means less crucial. Particularly, the designation of protected areas (PA) can be considered an (ecological) public function (Ring, 2002). Regarding this context, an innovative instrument has gained attention in recent years: Ecological Fiscal Transfers (EFT) change the redistribution of tax revenue by incorporating ecological indicators, for example, the existence of PA, into the fiscal transfer scheme. EFT have first been introduced in the Brazilian federal state of Paraná in 1992 and subsequently in 17 out of 27 Brazilian states (Droste et al., 2017c; Grieg-Gran, 2000;

Loureiro, 2002; Loureiro, Pinto, and Motta, 2008; May et al., 2002; Ring, 2008c; Vogel, 1997). Portugal has been the first state to introduce an EFT scheme on a national level in 2007 (Santos et al., 2012, 2015). From a theoretical perspective, EFT schemes have been proposed and simulated for Switzerland (Köllner, Schelske, and Seidl, 2002), India (Kumar and Managi, 2009), Indonesia (Irawan, Tacconi, and Ring, 2014; Mumbunan, 2011), Germany (Schröter-Schlaack et al., 2014), and France (Borie et al., 2014).

As such, EFT have a range of interesting features (Droste et al., 2017c): i) they may not require additional budget but change the existing fiscal revenue redistribution (Grieg-Gran, 2000; May et al., 2002; Ring, 2008c); ii) they can incentivize nature conservation and thereby increase the supply of an underprovided public good (Droste et al., 2017c; Grieg-Gran, 2000; May et al., 2002; Ring, 2008c); iii) they take local preferences and local knowledge into account since both in Brazil and Portugal they are general purpose transfer and responses are the choice of local decision makers;[33] iv) transaction costs for the introduction of EFT are relatively low because they constitute a rather marginal change in existing fiscal transfer schemes (Ring, 2008c; Vogel, 1997); and v) in the pioneering state of Paraná in Brazil, EFT even include criteria for the quality of PA management in the fiscal transfer scheme which may enhance not just quantity but also quality of conservation areas and measures (Loureiro, Pinto, and Motta, 2008). Regarding the outcomes of EFT, so far few studies have studied the effect of EFT on the designation of PA econometrically (see for example Sauquet et al, 2014).

Analyzing EFT in Brazil with an econometric panel data approach for $1991 - 2009$, Droste et al (2017) find evidence that introducing EFT creates an incentive effect for an additional designation of PA. They furthermore find indications for a decentralizing effect in the introduction of EFT since especially municipalities respond by designating additional PA. In general, decentralization provides means to incorporate local needs and preferences in polycentric and multilevel governance systems (Andersson and Ostrom, 2008; Faguet, 2014; Rubinchik-Pessach, 2005). In particular, decentralized conservation decisions can take into account relevant ecosystems that provide goods and benefits mainly to the local level but also conserve local habitat with endemic species and thus contribute to national and global conservation goals (Butchart et al., 2015; Smith et al., 2009). Hence, there are spill-over effects associated with local conservation action which can be internalized through a respective fiscal remuneration (Ring, 2008b). Given budgetary constraints for local governments, recognizing such spill-overs can change relative costs of provision and thus induce an incentive for an increased provision of local conservation. Focusing on the decentralization effect of introducing ecological indicators within fiscal transfers systems, we analyze the Portuguese EFT scheme as a case study for the first implementation of EFT that consider local governments' conservation policies within national level fiscal transfer schemes.

[33]For an analysis of strategic interactions at the local level in the Brazilian state Paraná see Sauquet et al, 2014

The Portuguese case may serve as a model for other countries and its effects on municipal PA designations thus embody policy relevance beyond the national scope.

Since 1993 municipalities in Portugal are formally permitted to designate their own PA and in 2008 a reform widened the range of municipal conservation competencies. In this context, we study whether the 2007 introduction of EFT in Portugal has incentivized municipalities to make use of their (enlarged) conservation competencies to designate PA, and in this sense, led to a decentralization of the decisions where to protect nature. Our research question therefore is: *Did the introduction of EFT in Portugal support the decentralization of conservation decisions, namely increase municipal PA designations in relation to national PA designations*? To this end, we employ the means of a Bayesian structural time series approach (Brodersen et al., 2015), which has the benefit of providing an estimated counterfactual time series for Portugal – simulating what would have happened without the intervention of introducing EFT; controlling for the simultaneous shift in nature conservation law. We are the first to assess the effect of a change in a fiscal governance regime on conservation planning outcomes through a Bayesian simulation of a counterfactual time series.[34]

The structure of the paper is as follows: section 4.4.2 introduces relevant literature on the theory of decentralization and fiscal federalism in relation to conservation governance; section 4.3 introduces a theoretical model of conservation decisions; section 4.4 provides background information on the relevant institutions in Portugal, namely the 2007 reform of the Local Finances Law that introduced the EFT scheme, and the conservation competencies of different governments to designate a range of PA categories, including the 2008 reform; section 4.5 gives the data sources and introduces the Bayesian structural time series approach; section 4.6 provides the results of analysis; section 4.7 gives the outcomes of robustness checks; in section 4.8 we provide methodological remarks on the quantitative approach; we discuss implications of results in section 4.9 and conclude briefly in section 4.10.

4.2 Literature Review – Decentralization and Conservation

The economic theory of fiscal federalism has its origins in the field of public finance (Musgrave, 1959; Oates, 1972, 2005). As an early scholar on the subject Friedrich von Hayek (1945) argued that decentralized systems provide informational advantages since local actors have more precise information of the needs, preferences and conditions of their 'immediate surroundings' than a central actor. According to Qian and Weingast (1997) this assumption refers to both consumers and local governments. Another important contribution was provided by Samuelson's theory of pure public

[34]For an application of voter behaviour with web search data see Street et al., (2015).

goods and public expenditure (Samuelson, 1954, 1955). For public goods where consumption is below national scale, say local public goods, local governments are assumed more efficient in providing the locally desirable level of output (Inman and Rubinfeld, 1997; Qian and Weingast, 1997; Tiebout, 1956) – given the absence of economies of scale (Olson, 1969). Because local constituencies may have different preferences and opportunity costs a local provision of regionally differentiable public goods maximizes welfare in comparison to a reference scenario of a central government providing an equal output level for all municipalities. Furthermore, the optimal level of provision of (local) public goods is also determined by the distribution of costs and benefits. Matching costs, benefits, and decision-making competencies was called the principle of 'fiscal equivalence', basically stating that for an optimal supply those who benefit from a provision of a public good should also bear the costs of provision, and therefore hold the competencies to decide on it (Olson, 1969).

These theoretical models have been generalized into a proposition known as the 'decentralization theorem' (Oates, 1972). However, since governmental structures cannot in every case coincide with the spatial coverage of the public good in question, interjurisdictional spillover effects may occur, e.g. by roads or clean rivers (Oates, 2005) or species conservation (List, Bulte, and Shogren, 2002). In such cases a fiscal transfer from a more central to a decentral governmental level can internalize such positive spillovers in the sense of a Pigouvian subsidy (Oates, 2005, see also Zodrow and Mieszkowski, 1986). Furthermore, it has been shown that even in the absence of informational asymmetries and a cheaper provision of particular public goods at a central government, decentralization can be beneficial in terms of welfare since projects of only local importance are realized (Rubinchik-Pessach, 2005).

These contributions on optimal allocations of costs and benefits among government levels assume, to a greater or a lesser extent, a welfare maximizing governmental behavior (Brennan and Buchanan, 1980; Feld, 2014). Thus, at all government levels the respective actors assumingly seek to promote the interest of their people (Oates, 2005). Such theory of optimal fiscal revenue allocation has been called the first generation fiscal federalism (Oates, 2005). Since the assumption of a welfare maximizing government might not always be fulfilled, a second generation fiscal federalism has been developed in order to analyze the 'black box' of governmental behavior (Qian and Weingast 1997). Drawing upon the theory of the firm (Coase, 1937), its updates, and public choice theory, Qian and Weingast, (1997) develop a theory of how governmental actors react upon institutional incentives and informational constraints. Oates (2005) extends the second generation theory of fiscal federalism to budget constraints, risk-sharing insurances and self-enforcing mechanisms in intergovernmental settings. Latest works include analyses of incentives and (de-)centralization tendencies (Weingast, 2009, 2014), decentral governance quality in general (Faguet, 2014), and in particular, the responsiveness of government spending to local needs (Borge, Brueckner, and Rattsø, 2014; Faguet, 2004). Local municipal actor involvement in national policy formulation has been analyzed regarding corresponding effects on successful implementation of those policies (Terman and Feiock, 2014), and in terms of causal relations of municipal spending and taxing behaviors to either locality bound micro incentives or institutional macro-level structures (Smith and Revell, 2016).

Observations of state-federal conflicts regarding environmental public functions have led to a general analysis and comparison of command and control, taxes and tradeable permits (Williams, 2012). Boadway and Tremblay (2012) identify environmental federalism and the governance of natural resource as unsolved challenges for future research and particularly name the organization of regulatory competencies, intergovernmental fiscal relations and incentive structures within multi-level governments as knowledge gaps. In this context, urban conservation behavior under budget constraints and the fiscal implications have been modeled from a micro-economic perspective (Wu, 2014) but without considering a multi-level structure.

We draw upon this body of literature, study municipal behavioral responses to fiscal incentives within multi-level government structures, and extend it into the direction of the provision of those public goods that are eminently supplied by protected areas (PA) – such as biodiversity conservation (Perrings and Gadgil, 2003; Ring, 2002, 2008b). PA are mostly designated at higher levels of government but management and opportunity costs related to these areas mostly occur at local levels. EFT compensate for foregone income, thus lower opportunity costs of hosting PA for local jurisdictions and potentially incentivize PA designations. Our argument is thus twofold. Firstly, EFT may compensate for management and/or opportunity costs at the local level that are incurred through the realization of (supra-) national conservation interests. Secondly, EFT may create an incentive for the designation of decentral PA through a change in conservation costs by a per area transfer for PA. This two-sided argument reads as follows.

On the one hand, there are national conservation interests, such as providing a high connectivity habitat network across the nation or the protection of large and nationally important sites through national parks. Furthermore, there even are supra-national interests such as the European Natura 2000 network that ensures a protection of important habitats and species across Europe. For these cases of overarching interests a central planning is better suited than a decentral implementation, since local decision makers are unlikely to consider these (supra-) national interests in their rationale unless fully internalized. Such internalization is difficult to realize since both opportunity and management costs of a habitat network may well differ across sites and regions and would thus require a spatially differentiated scheme in order to fully internalize the overarching interests in local decision making. EFT however, are generally lump-sum transfers that are not regionally differentiated. Through a uniform per area rate they may only (partially) compensate for opportunity costs incurred to the local level.

Such a (partial) compensation may nevertheless lower the resistance of local jurisdictions to PA planned and designated at higher levels of government.

On the other hand, most of the benefits from PA are of a regional nature, such as health, recreation and amenity services (ten Brink et al., 2013). Additionally, there are the positive spillover effects to the state, national and even the global level which originate from those services with a long spatial (and temporal) range such as climate regulation, biodiversity maintenance or water regulation (ten Brink et al., 2013). We assume that those services may not just be provided by national PA but also by local ones, but spatial spill-over benefits are often not internalized in local decisions. Since costs and also benefits differ among location and conserved habitats, a uniform EFT scheme would not internalize these positive external effects in a targeted manner but still create an incentive to increase decentral provision of PA. The incentive effect lies in the change in the relative prices (see section 4.3). If, for example, every per cent of a local jurisdictions territory that is put under protection receives a transfer quota, this reduces the price of providing local PA and therefore, likely yields additional local PA. Furthermore, this incentive effect would be greatest where the preferences for a local PA are largest and thus a change towards a positive net gain is most likely.

Considering these two sides of EFT, theoretically they lead to welfare gains by: a) reducing local costs through compensation for burdens incurred by centrally planned PAs, and b) better taking into account both local preferences and positive spatial externalities in the designation of smaller scale, local conservation areas.

4.3 Theoretical Model – Local Public Conservation Decisions

There can be a range of factors ultimately determining local decision maker's conservation spending and regulatory decisions. Starting from a neoclassical textbook definition, local decision makers could be considered as rational actors optimizing pay-offs corresponding to their preferences (or arguments of their utility function). It has been noted that such rationality is bounded by cognitive capacities (Simon, 1955), that commitment is a fundamental part of decision making (Sen, 1977) and that institutions define actions at least as much as intrinsic motivation (Ostrom, 1990; Vatn, 2007). Furthermore, local governments are no unitary actor but consist of multiple actors that all have their own agenda beyond the collectively defined one (Olson, 1965).

There is, however, no doubt that local governments face budget constraints. Let us consider a situation where a local decision maker has to decide between spending public budget on either conservation policy or some other public good out of all possible ones. The outmost boundary is given by the budget constraint, such that all available money income M is spent on either conservation action X at price p_x or a composite public good Y at price p_y (equation 4.1).

$$M = p_x X + p_y Y \tag{4.1}$$

Canonically, the optimal choice regarding quantities of X and Y is determined by both relative prices and marginal utilities U_x' and U_y', such that $\frac{p_x}{p_y} = \frac{U_x'}{U_y'}$. A policy that induces a price change, such a per unit fiscal transfer for PA, say from p_x to p_x', may lead to a greater quantity of conservation action, ceteris paribus (see section 4.4 for institutional details). For the sake of simplicity, let us assume that both goods are normal goods, and that there is some degree of substitutability between the two public goods. Then, $\Delta X = X(p_x', M) - X(p_x, M) > 0$ if $p_x' < p_x$.

From this simplistic model we would thus hypothesize that introducing fiscal transfers for PA leads to an increase in PA, given that PA spending leads to PA designations. There are, however, many more factors that determine the decision making of local government agents including various, right-based considerations, attitudinal beliefs and other intrinsic motivations beyond just monetary considerations (cf. works on factors determining willigness-to-pay for conservation: Kotchen and Reiling, 2000; Spash, 2006; Ojea and Loureiro, 2007; Spash et al, 2009). While such motivations may also alter the degree to which monetary considerations are taken into account in public administration and political conservation decision making, we would nevertheless base our analysis on the following simplifying hypothesis:

Hypothesis 4 *If designating PA becomes a source of income for local governments there will be an increase in corresponding conservation action.*

4.4 Institutions – Ecological Fiscal Transfers and Conservation Competencies in Portugal

While Portugal has a unitary government, there are some municipal and regional (fiscal) competencies, regarding e.g. taxation (Costa and Carvalho, 2013) or water management (Thiel, 2015). In this section we elaborate on the institutional context in Portugal concerning i) the introduction and functioning of EFT (section 4.4.1) and ii) the municipal and regional competencies in nature conservation, focusing on the designation of PA (section 4.4.2).

4.4.1 Ecological Fiscal Transfers

The Portuguese EFT were introduced through the Portuguese Local Finances Law (*Lei das Finanças Locais no. 2/2007*) reform in 2007 (Santos et al., 2012). The law establishes new rules for revenue distribution and fiscal transfers from central government funds to the local level, and was reformed again in 2013 but without a change in the EFT component (*Lei no. 73/2013 do regime financeiro das autarquias locais e das entidades intermunicipais*). On average total transfers account for about 44 per cent of total municipal income during $2007 - 2014$ (Direção-Geral das Autarquias Locais, 2015) while the rest is levied by municipal taxes on e.g. property, income and business (Santos et al, 2012, see also Costa and Carvalho, 2013).

There are three main national funds for disbursement of public revenue among municipalities. The Financial Equilibrium Fund (*Fundo de Equilíbrio Financeiro*) is a general grant with a value of 19.5 per cent of the arithmetic mean of income tax, corporate tax, and value added tax revenues (in 2007 it was 25.3 per cent). The Financial Equilibrium Fund is divided into two sub-funds with 50 per cent each, the General Municipal Fund (*Fundo Geral Municipal*) and the Municipal Cohesion Fund (*Fundo de Coesão Municipal*) for fiscal imbalances (Direção-Geral das Autarquias Locais, 2015; Santos et al., 2012). Moreover, there also is the Municipal Social Fund (*Fundo Social Municipal*) for expenditures on social public functions such as education, health and welfare. Additionally, a 5 per cent share of the income tax also goes directly to the municipalities (Direção-Geral das Autarquias Locais, 2015; Santos et al., 2012). Beyond those national funds, there are also transfers from the European Union to municipalities (Direção-Geral das Autarquias Locais, 2015).

With regard to EFT the funds' allocation works the following way. Among other criteria, 5 per cent of the General Municipal Fund (GMF) are allocated in proportion to the area under protection (Natura 2000 and other PA). In case more than 70 per cent of the municipal area is under protection the ecological component portion becomes 10 per cent – which reduces the otherwise 25 per cent of the GMF redistributed according to area to 20 per cent (*Lei no. 2/2007* and *Lei no. 73/2013*). This makes EFT 2.5 to 5 per cent of the Financial Equilibrium Fund. It is important to note, however, that the EFT are general purpose transfers without any earmarking. While the allocation of EFT in Portugal is based on the existence (and expanse) of PA, the municipalities can spend the respective income on whatever public function they consider necessary. Figure 4.1 gives an overview of the structure of fiscal transfers in general and the EFT in particular.

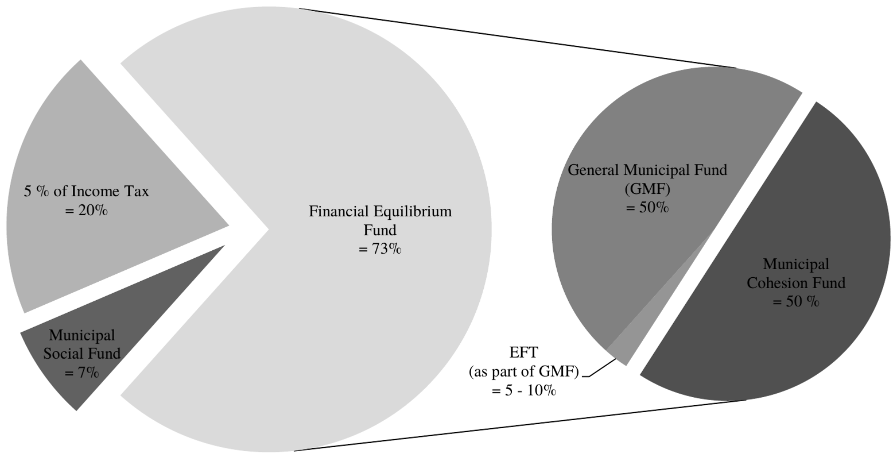

FIGURE 4.1: Fiscal transfer funds and EFT in Portugal. *Source:* authors' elaboration based on data from Direção-Geral das Autarquias Locais (2015). The left bubble represents the 2015 distribution funds (tax income varies). The right bubble is defined by law (both general and cohesion fund are always 50 per cent of the equilibrium fund).

Analyzing the transfers based on the 2007 reform Santos et al. (2012, p. 271) compare the reform to a simulated computation that excludes the ecological component. They derive that EFT have an average unit value of 50 €/ha PA for municipalities with more than 70% of PA on their territory and 25 €/ha for those municipalities with less PA. Calculating the EFT proportion of total municipal revenues and fiscal transfers they find a range of 4% to 38% for the group with at least 70% PA and on average less than 8% for the remainder. This is to say, the average EFT unit values are rather small over all but incentives for some municipalities can be quite strong.

4.4.2 Nature Conservation Competencies

The competencies regarding the designation of PA in Portugal are divided between the national, the local, and the private level. In Table 4.1 there is an overview of different PA categories per government level. We briefly introduce each in turn.

The European Natura 2000 network site selection is based on lists of ecologically important natural habitats and species, known as Sites of (European) Community Importance (Evans, 2012). Based on these lists the Portuguese national authorities decide upon the designation of Special Areas of Conservation (SAC) under the Habitat Directive and the Special Protection Areas (SPA) under the Birds Directive.

The national authorities (i.e. the Environmental Ministry and its agency, the Institute for Nature Conservation and Forests (Instituto da Conservação da Natureza e das Florestas – ICNF) can designate all IUCN (International Union for the Conservation of Nature) PA categories such as national parks, nature parks, nature reserves, protected landscapes areas and nature monuments (*Decreto Lei no. 142/2008*).

TABLE 4.1: Protected area designation competencies of different governmental levels and their legal foundation

Designating body	PA categories	Legal foundation
National authorities	– Special Area of Conservation – Special Protection Area	– EU Habitats Directive – EU Birds Directive
	– National Park – Nature Park – Nature Reserve – Protected Landscape Area – Nature Monument	– Decreto-Lei n.o 19/93 – Decreto-Lei n.o 142/2008
Regional and local municipal authorities	– Nature Park – Nature Reserve – Protected Landscape Area – Nature Monument	– Decreto-Lei n.o 19/93 – Decreto-Lei n.o 142/2008
Private landusers	– Private PA	– Decreto-Lei n.o 142/2008

Source: authors' elaboration based on ICNF (2015), see also Santos et al., (2012)

The municipalities or regional associations of several municipalities may designate all these PA categories except national parks. It is important to note that while the law decree 19/1993 defined that municipalities and municipal associations can *propose* the designation of only a regional protected landscape area to the ministry, the law decree 142/2008 widened their competencies and authorizes them to directly *designate* all PA categories but a national park. However, out of the eight regional and local PA designated on basis of law decree 142/2008 only three are not protected landscapes areas, meaning there are relatively few responses to the 2008 widening of municipal PA designation competencies regarding the type of designated PA (see table **??** in the appendix for some detail). In practice, the change from proposing a local PA and designating it at the local level can be considered a fairly slight change since even under the 2008 regime, official recognition of municipal PA designations is subject to Ministerial decision. Furthermore, the 2008 reform allowed explicitly for the designation of private protected areas. So far, there is one private PA (Faia Brava). With the exception of Natura 2000 sites, these protected sites altogether constitute the national network of protected areas (Rede Nacional de Áreas Protegidas – RNAP).[35]

4.5 Empirics – Bayesian Structural Time Series Analysis

4.5.1 Data

Focusing on the PA designated under Portuguese law, we collected data on designated protected areas from the Institute for Nature Conservation and Forests (ICNF) that account for the national network of protected areas but do not include Natura 2000 areas except those parts that are spatially overlapping with the national PA (ICNF, 2015), socio-economic controls representing the general structure of the economy such as GDP per capita, population density, value added by the agricultural, industrial, and service sectors from the World Bank (2015), and controls representing conservation preference proxies such as data on members of environmental NGO per 1, 000 inhabitants, municipal spending and income related to the environment (regarding climate and air quality, waste water treatment, residual waste treatment, water protection, noise reduction, biodiversity and landscape protection, radiation control, research and development and other environmental protection) from the National Statistics Institute (INE, 2015). All monetary values are given in constant €2005 prices.[36] This way, we constructed a multivariate time series for Portugal from 1995 to 2014 with yearly observations. Summary statistics and time series of PA data can be found in the appendix, and the compiled raw data and code for reproducing results is provided in a personal github repository (*in order not to violate anonymity link is to be inserted later*).

[35] In this context, it is worth noting, that the Natura 2000 network (including most other PA) covered 18.8 per cent of continental Portugal in 2010, while the RNAP only accounted for 7.9 per cent (INE, 2015) and in 2013 Natura 2000 covered 20.7 per cent of entire Portugal (EU, 2015) while the RNAP accounted for 8.5 per cent of Portugal (ICNF, 2015). This is due to the special nature of Natura 2000 sites which are not necessarily to be designated as PA under national law but managed according to EU law. The EFT mechanism, however, accounts for both Natura 2000 and RNAP sites.

[36] Monetary values were deflated based on the World Bank GDP deflator for Portugal or calculated in Euro with average US dollar exchange rates for 2005.

4.5.2 Econometric model

Since we want to estimate the effect of the 2007 EFT introduction on the degree of centrality in conservation decisions, measured by the ratio of municipal and national PA designations, we employ a model constructing an appropriate counterfactual via a synthetic control. The ***CausalImpact*** package (Brodersen et al., 2015) within **R** (R Development Core Team, 2016) provides such an implementation by employing a Bayesian structural time series approach. Originally designed to infer effects of online marketing interventions, ***CausalImpact*** estimates the post intervention difference between the observed time series of the response variable and a simulated (synthetic) time series that would have occurred without the intervention (Brodersen et al., 2015). The posterior causal inference functions the following way: The model is first estimated with the pre-intervention data. Then the dependent variable is predicted over the post-intervention period using the observed value of the explanatory variables. The difference between the prediction and the observed values of the dependent variable during the post-intervention period is interpreted as the impact of the policy intervention. The counterfactual post-intervention prediction is thus basically built through three sources of information: i) the dependent time series behavior prior to invention, ii) covariate time series pre-intervention behavior with predictive power for the response variable time series, and iii) if existent, available prior knowledge about the model parameters since it is a Bayesian framework (Brodersen et al., 2015).

The employed Bayesian structural time series model is a state-space model for time series data which can generally be defined as a pair of equations:

$$y_t = Z_t^T \alpha_t + \epsilon_t \tag{4.2}$$

$$\alpha_{t+1} = T_t \alpha_t + R_t \eta_t \tag{4.3}$$

where $\epsilon_t \sim N(0, \sigma_t^2)$ and $\eta_t \sim N(0, Q_t)$ are error terms independent of all other unknowns (Brodersen et al., 2015). Equation 4.2 is the *observation equation* where the response variable y_t is linked to a d-dimensional state vector α_t and an independent and identically, normally distributed error term ϵ_t. $Z_t \in \mathbb{R}^d$ denotes an output vector. Equation 4.3 is the *state equation* that covers the behavior of state vector α_t. Here, the matrices $T_t \in \mathbb{R}^{d \times d}$ and $R_t \in \mathbb{R}^{d \times q}$ are transition and control matrix respectively, where $q \leq d$, and $Q_t \in \mathbb{R}^{q \times q}$ denotes the state-diffusion matrix of the above mentioned system error $\eta_t \in \mathbb{R}^q$, see Brodersen et al (2015, p. 252). In our case, we estimate the basic local level model with contemporaneous covariates and with static, that is time-invariant coefficients. This can be achieved by setting $Z_t = \beta^T x_t$ and $\alpha_t = 1$ (Brodersen et al., 2015). In order to account for local variation in time series we also specify a local linear trend model (see equations 4.4 and 4.5) for the robustness tests (see section 4.7). The local linear trend can be be defined by the pair of equations:

$$\mu_{t+1} = \mu_t + \delta_t + \eta_{\mu,t} \tag{4.4}$$

$$\delta_{t+1} = \delta_t + \eta_{\delta,t} \tag{4.5}$$

where $\eta_{\mu,t} \sim N(0, \sigma_\mu^2)$ and $\eta_{\delta,t} \sim N(0, \sigma_\delta^2)$ (Brodersen et al., 2015). Parameter μ_t represents the local trend of the response variable at time t, and δ_t corresponds to the change in μ between t and $t+1$ or, in other words, the slope at time t exhibits a random walk (Brodersen et al., 2015).

Semi-local trend models (which we also use as a robustness check in section 4.7) are more useful for estimating long-term predictions since the slope is modeled as stationary AR(1) process instead of a random walk which makes it less variable (Brodersen et al., 2015). The model can be expressed as

$$\mu_{t+1} = \mu_t + \delta_t + \eta_{\mu,t} \tag{4.6}$$

$$\delta_{t+1} = D + \rho(\delta_t - D) + \eta_{\delta,t}, \tag{4.7}$$

where $\eta_{\mu,t}$ and $\eta_{\delta,t}$ are independent, the slope of the time trend varies with an AR(1) process around the long-term slope of D which is estimated with a a Gaussian prior, and $|\rho| < 1$ is the learning rate of the local trend updates which is estimated with a Gaussian prior truncated to (-1, 1) (cf. Brodersen et al., 2015).

We employ this Bayesian structural time series framework to estimate the effect of EFT introduction (our intervention starting in 2007) on the ratio of municipal and national PA. We have chosen the ratio of municipal and national PA in order to account for the degree of decentrality in conservation decisions. The higher the ratio, the more municipal PA there are in relation to national PA. If the ratio significantly increases after EFT were introduced, this would indicate a decentralizing effect of transfers (for PA provisions) from state to the municipal level. For the robustness checks (see section 4.7), however, we also account for the area covered by PA in order to account for area-wise decentralization effects in PA designations.

The socio-economic covariates' time series, namely GDP per capita, population density, value added by each the agricultural, industrial, and service sectors, members of environmental NGOs per 1,000 inhabitants, and municipal spending and income related to the environment (for data sources see section 5.1) are included according to a spike-and-slab prior of the predictors. The spike places a positive probability mass at zero for the coefficients, the slab poses a weakly informative prior parameter distribution through a close to flat Gaussian with large variance, and the models include nonzero predictors (Scott and Varian, 2014). The spike-and-slab prior ensures that sparse models with few but powerful predictors are estimated. The model algorithm chooses an appropriate set of covariates within a forward-filtering, backward-sampling framework, based on a Kalman filter. The filter recursively computes the predictive distribution $p(\alpha_{t+1}|y_{1:t})$ moving forward through the time series, while the Kalman smoother moves backward through time updating the output of the Kalman filter (Scott and Varian, 2014). The algorithm averages the final model over parameter value results of a Markov chain Monte Carlo (MCMC) simulation of several model draws that are each based on the spike-and-slab prior and thereby include different (sub-)sets of controls (Brodersen et al., 2015; George and McCulloch, 1997; Scott and Varian, 2014). In our case we set the number of MCMC model draws to $10,000$. The model structure with a Bayesian model averaging over models based on a spike-and-slab prior allows for uncertainty in model-selection while we can report both the marginal probability with which particular co-variates were included, thus on their predictive power, and the marginal probability of e.g. a positive coefficient.

4.6 Results – Decentralization effects in Portuguese Ecological Fiscal Transfers

During the post-intervention period, namely after the introduction of the EFT, the response variable, that is to say the ratio of municipal and national PA, had an average value of approximately 0.30. By contrast, in the absence of the intervention, we would have expected an average response of 0.14 with a 0.02 standard deviation (SD). The 95% confidence interval (CI) of this counterfactual prediction is $[0.10, 0.18]$. Subtracting this prediction from the observed response yields an estimate of the causal effect the intervention had on the response variable. This effect is 0.17 with a 0.02 SD and 95% CI of $[0.13, 0.21]$. This means that if we predict the development of the ratio of municipal and national PA numbers during the postintervention period, given the pre-intervention period correlations of the control variables and the post-intervention development of these variables, the observed ratio is about 0.17 higher than we would have expected.

Summing up the individual data points during the post-intervention period, estimating a cumulative impact, the response variable of the ratio of municipal and national PA counts had an overall value of 2.10. By contrast, had the intervention not taken place, we would have expected a sum of 1.16 with a SD of 0.15 and a 95% CI of $[0.88, 1.45]$. The above results are given in terms of absolute numbers. In relative terms, the response variable showed an increase of $+120\%$ with a SD of 16% . The 95% CI of this percentage is $[+91\%, +150\%]$. The probability of obtaining this effect by chance is very small (Bayesian tail-area probability $p = 0.0001$). This means that the positive effect observed during the intervention period is statistically significant and unlikely to be due to random fluctuations. Summarizing, our estimation shows that the ratio of municipal and national PA numbers has significantly increased after EFT were introduced in Portugal, which we infer to be a consequence of the fiscal incentive effect that is inherent in designating a percentage of tax income transfers to municipalities according to ecological criteria (see section 4.9.1 for a further comment on causal inference). For a graphical illustration of our analysis see Figure 4.2.

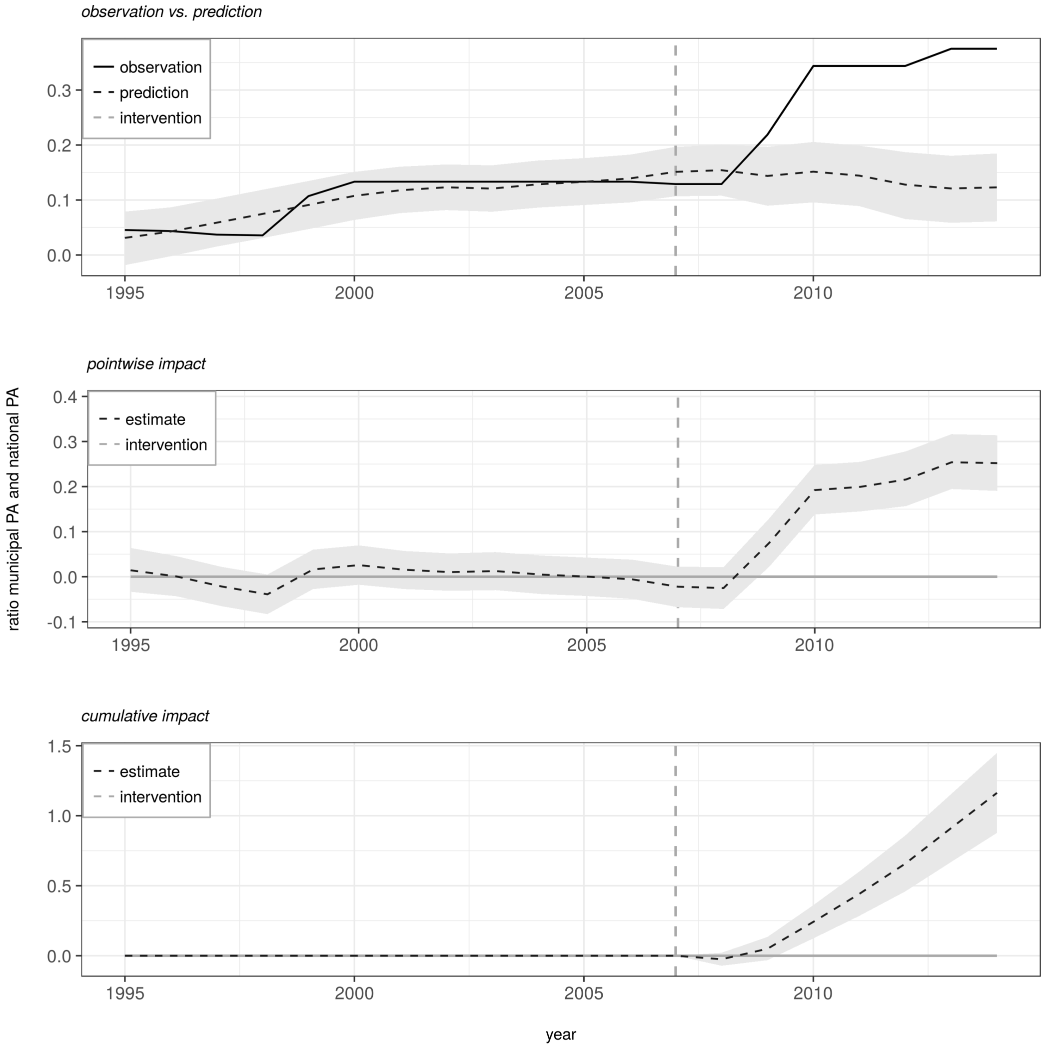

FIGURE 4.2: Graphical illustration of Bayesian Structural Time Series model results: i) observation vs. prediction, ii) pointwise impact and iii) cumulative impact estimates, all with grey shade as uncertainty range between upper and lower limit estimates. Note that the uncertainty range in the post-intervention period slowly increases from 2008 to 2014: for the prediction from $[0.108, 0.201]$ in 2008 to $[0.061, 0.185]$ in 2014, for the pointwise impact from $[-0.072, 0.021]$ in 2008 to $[0.190, 0.314]$ in 2014, and for the cumulative impact from from $[-0.072, 0.021]$ in 2008 to $[0.877, 1.448]$ in 2014. *Source:* authors' computation.

Figure 4.3 displays the marginal posterior inclusion probability of control variables. This gives insight into how the different model draws are structured and about the average probability of the sign of coefficients. It shows that GDP per capita is by far the most predictive covariate with regard to the ratio of municipal and national PA numbers and has a positive sign on average. With less predictive power and a slightly lower probability of a positive sign, follow population density and value added by the service sector (which accounts for a large portion of GDP per capita) and even less so value added by the industry sector. Members of environmental NGO per $1,000$ inhabitants most probably has a negative sign. The other covariates have a low probability of a positive sign and a relatively low predictive power since they are rarely included in models drawn from $10,000$ simulated models. The Monte Carlo Standard Errors of the estimated coefficients can be found in table 4.4 in the appendix.

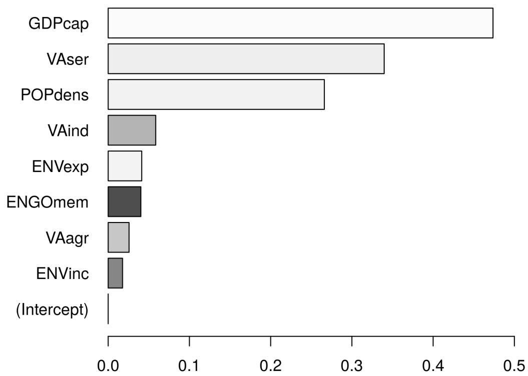

FIGURE 4.3: Marginal posterior inclusion probability of variables in 10,000 model draws. Color shades are in proportion to the probability of a positive coefficient on a continuous $[0, 1]$ scale: negative coefficients are black, positive coefficients are white, and gray indicates an indeterminate sign, of a probability of a positive coefficient around 0.50) (Scott and Varian, 2014). Variables are (with probability of a positive coefficient in parentheses): *GDPcap* is GDP per capita (0.99), *POPdens* is population density (0.95), *VAser* is value added by the service sector (0.94) *VAind* is value added by industrial sector (0.71), *ENVexp* is the environmental municipal expenditure (0.95), *ENGOmem* are members of environmental NGOs per $1,000$ inhabitants (0.31), *VAagr* is the value added by agricultural sector (0.78), and *ENVinc* is the environmental municipal income (0.53). *Source:* authors' computation.

4.7 Robustness Checks

Because there was a reform of the PA designation competencies in 2008, just after the introduction of EFT in 2007, we conduct a robustness check for this second and almost simultaneous regime shift. This first check consists in excluding those three municipal, respectively regional municipal association, PA designations (in terms of the number of PA designated) from our data set that have only been possible with the reform of the nature conservation competencies (the regional Natural Park Vale do Tua, the local Natural Reserve Estuário do Douro, and the local Natural Reserve Paul de Tornada). The result shows that the estimated causal effect would be lower than in our initial analysis but it would still be significant. The difference between observed post-intervention average of the response (0.24) and the prediction without an effect (0.14 with a 0.02 SD and a 95% CI of $[0.09, 0.18]$) would be 0.10. In relative terms, the response variable showed an increase of 76% (15% SD). The 95% interval of this percentage is $[+47\%, +104\%]$. The Bayesian tail-area probability is $p = 0.0001$. Hence, even when excluding those above mentioned PA designations, that have only been possible with the law decree $142/2008$ reform of municipal nature conservation competencies, we consistently estimate a significant causal effect with this model.

Since national-level PA are on average larger in size than municipal PA and we are interested in the decentralization effect regarding PA designations we measured the dependent variable as the ratio of municipal and national PA. However, conservation is not just about the number of PA but also about their expanse. Therefore, as another robustness check, we repeat the analysis with the respective ratio of the area in hectares of municipal and national PA (see table **??** in the appendix for some detail). The results show a similar but weaker effect – which is due to the difference in sizes of municipal and national PA. The observed post-intervention average of the response is 0.027 and the counterfactual prediction 0.015 with a 0.002 SD and a 95% CI of $[0.009, 0.019]$ which means an estimated effect of 0.012 (SD 0.002, CI $[0.008, 0.018]$). In relative terms, the results are comparable to the analysis of PA numbers: the response variable showed an increase of $+83\%$ (SD 16% and a 95% CI of [+52%, +123%]). The Bayesian tail-area probability is $p = 0.0001$.

However, if we proceed to estimate the EFT effect on the ratio of municipal and national PA in terms of area covered and also exclude those municipal PA that have only been possible with the 2008 reform of municipal conservation competencies, there is no longer a significant effect. Predicting a counterfactual response of 0.015 (SD 0.0024, 95% CI $[0.009, 0.019]$) we fail to reject the null hypothesis of a significant difference to the observed ratio of 0.017. This is due to the fact that the area-related effect is mainly driven by one singular municipal PA, namely the regional Natural Park Vale do Tua. This regional park has been designated in 2013 and, with a size of about $24,767$ hectares, is comparable to the size of national PA. It has furthermore been designated by a regional association of the municipalities of Alijó, Murça, Vila Flor, Carrazeda de Ansiães, and Mirandela (ICNF, 2015). This means that in terms of PA area, the introduction of EFT in Portugal had no significant effect *without* the widening of municipal conservation competencies (see section 4.9.2 for a discussion of the relevance of compentencies for the functioning of the instrument).

Furthermore, we also computed (semi-)local trend models in order to account for local variation in time series. The local trend model is based on a random walk slope and the semi-local trend model is based on a stationary AR(1) process around a long-term trend (see section 5.2 for details). The local trend model has a strong variability which means long-term predictions may suffer from wide uncertainty intervals but the semi-local trend model balances short-term information with longer-term information from the past (Brodersen et al., 2015). As expected, the significance in the local trend model ceases: the predicted counterfactual response of 0.015 (SD 0.11, 95% CI $[-0.11, 0.34]$) is not significantly different to the observed ratio of 0.3. This means that, although the intervention appears to have caused a positive effect, this effect is not statistically significant when considering the entire post-intervention period as a whole. There appears to be significant effect for about a three year period after the intervention. However, the apparent effect could be the result of random fluctuations that are unrelated to the intervention. This can be the case when the intervention period is very long and includes much of the time when the effect has already worn off – as appears to be the case around 2010. To the contrary, estimating a semi-local trend model that exhibits less variation, we find a significant impact: the actual observation of 0.3 is significantly higher than the predicted counterfactual of 0.014 (SD 0.071, 95% CI $[-0.011, 0.28]$) with a Bayesian one-sided tail-area probability of $p = 0.014$. This means that, if we allow for great variation in our predictions the signifance ceases, potentially due to a wearing off effect, but the results are robust to allowing for a considerable amount of variation through an AR(1) process around a long-term trend.

4.8 Methodogical Remarks

Regarding our quantitative approach, the application of a method originally designed for assessing causal impacts of marketing interventions to the introduction of an economic instrument for nature conservation such as EFT, produced interpretable and sensible results. This mainly is a consequence of a neat implementation of the ***CausalImpact*** package within the **R** environment and the merits of the Bayesian framework. While the spike-and-slab prior allowed obtaining relatively sparse but predictive models, the MCMC simulations allowed a model averaging regarding the inclusion of the most predictive covariates. Building upon these algorithms, predicting a counterfactual time series is the key feature of the Bayesian structural time series approach. Thereby it provides a solution to a fundamental and long standing issue in econometric analysis of causal effects, the problem of not having a controlled experimental setting in analyzing real world phenomena or policies (cf. Ashenfelter and Card, 1985; Athey and Imbens, 2015; Box and Tiao, 1975; Heckman, 2008; Meyer, 1995).

Another statistical issue is the required length of the time series. While, for example, Box et al (2016, p. 15) state that long time series of about 50 to 100 observations are required for proper analysis, i.e. for data with seasonal variability, Simonton (1977) argues that for cross-sectional analyses time series with 4 to 12 observations per case can suffice. Hyndman and Kostenko (2007) state that it is at least required to have more observations than parameters but also differentiate between requirements for standard time series analysis methods such as regressions with seasonal dummies, Holt-Winters Methods and ARIMA models. They suggest a Bayesian framework for cases in which data is limited – which applies to our case. Furthermore, requirements may reduce if regularizing methods are applied (Hyndman, 2014) – like the spike-and slab prior in our model. This is to say requirements very much depend on the data structure, the nature of the observed variables and the modeling approach. In our data set we have yearly data of the dependent and 8 independent variables for the 13 years of the pre-intervention period, and 7 years for which the dependent variable is both observed and predicted as a counterfactual based on the pre-intervention correlations and post-intervention variability of covariates using regularizing priors (Brodersen et al., 2015). There is no seasonal variability in the dependent variable and a rather stable (close to linear) trend with a small jump around 1998-2000, and apparently rather static coefficients in the pre-intervention period without much random variation in the development of our dependent variable over time. Thus, we assume that a Bayesian prediction of a simulated counterfactual time series that takes into account the known post-intervention variation of covariates suffices for a reliable post-intervention estimation, especially since the technique has particularly been developed for short time series forecasting (Scott and Varian, 2014).

One potential shortcoming of the model is that covariates such as GDP per capita, population density and value added are potentially endogenous such that designating municipal PA may attract investments or inhabitants through local amenities. In general, such endogeneity may affect our results. However, there are two reasons why we assume those to be negligible, one circumstantial and the other methodological: i) considering the circumstances of the banking and fiscal crisis starting 2007, the relative importance of municipal PA for GDP and value added appears to be minimal and population movement might be more affected by employment than by local amenities; ii) while designating a municipal PA takes immediate effect, the change in habitat structure and quality and thus local amenities through conservation action require longer time, such that the prediction based on past and contemporaneous covariates would not be affected by lagged effects of the dependent variable on covariates.

Furthermore, it is important to note that something else could have happened after 2007 that also affected the PA designations, but that it remains an unaccounted for phenomenon which it is not captured in the set of controls. We account for economic variables over the 2008 crises, proxies for environmental values in the population such as environmental NGO members, and simultaneous institutional shifts. We are thus confident that we account for the most important variables. Yet, we cannot exclude the possibility of unobserved effects with certainty. There may have been some sudden shift in local decision makers' behaviour and PA designation acts that remains unexplained by our model.

4.9 Discussion – Motivations, Municipal Competencies, and Welfare Gains

Quantitative results show that the introduction of EFT is followed by an increase in the ratio of municipal and national PA numbers. While national authorities keep designating PA, municipalities designate more of their own PA categories than previous to the introduction of the scheme such that the ratio rises. We can observe a synchronicity of events in the time series, where the rise in the ratio of municipal and national PA coincides with the introduction of EFT in Portugal. Through a comparison of the post-intervention ratio with a simulated counterfactual time series predicted from pre-intervention correlations, we can infer the quantitative effect of an introduction of EFT. Given the Bayesian structural time series approach, these results suggest decentralization in nature conservation decisions through EFT.

For the discussion of these results we focus on three specific aspects: i) a note on motivational aspects of conservation decisions and causal inference, ii) municipal nature conservation competencies and their importance for the functioning of EFT, and iii) welfare implications of decentralization through EFT.

4.9.1 Motivations of Local Decision Makers to Designate Protected Areas

As a first remark, it is important to note that we observe outcome variables which are the result of the decisions on the local level but not the decision making process itself. As briefly introduced in the theoretical model section (see section 4.3), there can be a wide variety of actual reasons for designating municipal PA, among which the financial incentive inherent in EFT schemes may be found. Nevertheless, we can observe a synchronicity in the events of introducing EFT and a rise in the ratio of municipal and national PA. Given our theoretical proposition that fiscal remuneration incentivizes the designation of municipal PA and this synchronicity, we would argue that the outcome of decisions for municipal PA is thus a consequence of introduction of EFT. The inference of such a causal effect is however limited to a quantitative perspective. A qualitative analysis of motivations of those municipalities that have actually designated more PA after the introduction of the EFT scheme could identify and scrutinize the underlying decision-making processes and remains a task for future research.

4.9.2 Conservation Competencies and their Importance for the Functioning of Ecological Fiscal Transfers

Our results regarding a decentralizing effect of introducing EFT are robust to the exclusion of the *number of PA* designations that only have been possible with the 2008 reform of municipal nature conservation competencies. The effect is also significant if we measure outcome in terms of *PA area covered*. However, if we simultaneously measure area covered and exclude those municipal PA that were only possible with the 2008 institutional change the effect ceases to be significant (see section 4.7 for an explanation). This is to say, the incentive appears to work on both the number of designated municipal PA and the extent of those, but the latter would not have been possible without the simultaneous enlargement of municipal designation competencies. Hence, the effects that EFT can have on the designation of municipal PA in Brazil (Droste et al., 2017c) or in Portugal can very likely not be replicated in other countries unless there are comparable nature conservation competencies in place for the designation of municipal PA. The robustness checks thus indicate the importance of municipal competencies for the functioning of the instrument. They are a crucial element for the decentralization effect through EFT incentives and particularly important if only the extent of municipal PA was to be considered relevant for conservation effectiveness.

4.9.3 Welfare Implications of Decentralization through Ecological Fiscal Transfers

The fiscal remuneration of ecological public functions likely have welfare related effects (see section 4.4.2 for the theoretical underpinning) such as: a) a compensation of costs incurred to the local level through (supra-)national PA designations; b) the decentralizing effect which allows to take local conservation preferences better into account; and c) the increase in the provision of an undersupplied public good. Although we can observe an increase in local PA which we associate with the introduction of EFT, the welfare implications are based on the theory of fiscal federalism. The following remarks thus remain at a conceptual level.

As stated initially, decisions on the EU Natura 2000 network, or nationally important conservation sites, are reasonably better informed at the (supra-)national level where there are well-trained conservation experts with knowledge on the distribution of e.g. endangered species or important corridors for overarching habitat networks. For these (supra-)national PA designations, EFT can (partly) compensate for the costs imposed to the local level and thus reduce negative external effects of higher government level conservation planning. That municipalities have competencies to designate their own PA, however, opens a leeway for an incentive effect beyond mere compensation. Apparently, the relatively low unit value of EFT sufficed to incentivize the designation of additional municipal PA. Given that local decision makers indeed designate PA in (better) accordance with interests at the local level, the EFT induced designations would lead to more precise and locally differentiable preference satisfaction in the decision where to protect nature.

Furthermore, the additional municipal PA increase the provision of undersupplied local public goods such as biodiversity conservation with their potentially long spatial and temporal range spillover benefits. At the same time, however, there may be economies of scale in conservation (Armsworth et al., 2011). It is thus important to recognize that the introduction of EFT does not contradict or substitute but supplement conservation competencies of (supra-)national bodies such as central planning agencies or the European authorities. Thus, based on assumptions of the theory of fiscal federalism, EFT may yield welfare gains through i) compensating municipalities for costs incurred by (supra-)national conservation planning, ii) incentivizing decentral PA designations which are potentially in line with local preferences and iii) increasing the provision of undersupplied public goods and services through small to medium scale PA without counteracting (supra-)national large scale designations. These assertions, however, require more thorough welfare analyses – which remains a future task.

4.10 Conclusion

Analyzing the effect of the 2007 introduction of EFT in Portugal, we provide quantitative evidence of an increase in the ratio of municipal and national PA numbers in the post-intervention period. Comparing a simulated counterfactual time series, obtained by predicting pre-intervention correlations of socio-economic control variables with the observed outcome variable for the post-intervention period, we find a significant difference between counterfactual predictions and actual observations. We can thus observe a synchronicity of introducing EFT and the rise in the ratio of municipal and national PA, that is unlikely a consequence of random processes. Against the theoretical background, where we model how fiscal incentives may increase the designation of decentral PA through lowering relative prices, this observed decentralization effect has very likely been caused by the Portuguese EFT introduction.

Deducing implications from the theory of fiscal federalism, such decentralization may lead to welfare gains since local preferences could better be taken into account and spatial conservation spill-over effects from municipal PAs are (partially) internalized. At the same time, such an additional decentralization effect does not exclude a centrally planned designation of protected areas of (supra-)national importance, as the municipal competencies do not substitute but supplement conservation competencies of (supra-)national bodies. For such central PA designations the EFT compensates for costs imposed to the local level. Recognizing ecological public functions within fiscal transfers schemes thus has the potential to increase overall performance of the public sector.

An important implication of our analysis is how crucial municipal competencies for the designation of PA are for the decentralizing incentive effect – especially when considering ecological effectivenes in terms of PA coverage. Without those competencies municipal bodies would have no means to directly react to the incentive effect and increase the municipal supply of protected areas. As a response to the (inter-)national demands for biodiversity protection, introducing fiscal incentives through EFT has the potential to increase the likelihood of decentral conservation action, even without the need for additional expenditure, but only if decentral governments have corresponding conservation competencies.

Acknowledgments

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Appendix. Descriptive Statistics

Summary statistics

TABLE 4.2: Summary statistics

Statistic	N	Mean	SD	Min	Max
ratio municipal PA / national PA (*PAratio*)	20	0.173	0.117	0.036	0.375
value added by agriculture (*VAagr*)	20	3,711,254,633	136,704,563	3,566,115,527	4,110,711,457
value added by industry (*VAind*)	20	32,371,423,673	2,524,840,175	27,799,037,951	35,769,016,793
value added by service (*VAser*)	20	96,957,621,027	9,520,538,122	77,956,243,299	108,038,023,839
GDP per capita (*GDPcap*)	20	14,603.360	896.900	12,383.830	15,636.750
population density (*POPdens*)	20	113.517	1.958	109.576	115.439
municipal environmental spending (*ENVexp*)	20	573,327.100	53,616.450	468,352	663,297.500
municipal environmental income (*ENVinc*)	20	210,387.300	47,144.360	134,958	304,035.400
environmental NGO members per 1,000 inhabitants (*ENGOmem*)	20	4.600	2.280	1	8

Source: authors' calculations based on ICNF (2015), World Bank (2015), and INE (2015); monetary values are in constant €2005 prices.

Time Series of Dependent Variable Components

TABLE 4.3: Time series of PA designation variables at different government levels

Year	National PA area [ha]	Municipal PA area [ha]	Number of national PA	Number of municipal PA	N° municipal PA via 2008 reform
1995	623.360	3.282	22	1	
1996	623.414	3.282	23	1	
1997	623.434	3.282	27	1	
1998	710.435	3.282	28	1	
1999	710.435	10.360	28	3	
2000	742.191	10.706	30	4	
2001	742.191	10.706	30	4	
2002	742.191	10.706	30	4	
2003	742.191	10.706	30	4	
2004	742.191	10.706	30	4	
2005	742.191	10.706	30	4	
2006	742.191	10.706	30	4	
2007	742.309	10.706	31	4	
2008	742.309	10.706	31	4	
2009	743.274	11.206	32	7	2
2010	743.274	13.418	32	11	2
2011	743.274	13.418	32	11	2
2012	743.274	13.418	32	11	2
2013	743.274	38.185	32	12	3
2014	743.274	38.185	32	12	3

Source: authors' calculations based on ICNF (2015)

Monte Carlo Standard Errors

TABLE 4.4: Monte Carlo Standard Errors of estimated coefficients

	Base Model (BM)			**BM excluding 2008 reform PA**			**BM on area of PA**		
Variables	*MCSE*	*SD*	*MCSE /SD in %*	*MCSE*	*SD*	*MCSE /SD in %*	*MCSE*	*SD*	*MCSE /SD in %*
VAagr	0.0004	0.0370	0.89	0.0004	0.0410	0.99	0.0010	0.0580	0.98
VAind	0.0010	0.1220	1.01	0.0010	0.1150	1.07	0.0020	0.1640	1.15
VAser	0.0150	0.4870	3.19	0.0140	0.4720	3.03	0.0100	0.4620	2.23
GDPcap	0.0190	0.5150	3.66	0.0170	0.4900	3.45	0.0130	0.5340	2.44
POPdens	0.0130	0.4060	3.35	0.0140	0.4040	3.45	0.0060	0.2850	2.23
ENVexp	0.0010	0.0520	0.91	0.0010	0.0530	1.34	0.0010	0.0610	1.66
ENVinc	0.0010	0.0610	0.80	0.0010	0.0750	1.62	0.0010	0.0770	1.91
ENGOmem	0.0004	0.0390	0.81	0.0010	0.0540	1.61	0.0010	0.0530	1.74
	BM on area and excluding 2008 reform PA			**BM + local trend**			**BM + semi-local trend**		
Variables	*MCSE*	*SD*	*MCSE /SD in %*	*MCSE*	*SD*	*MCSE /SD in %*	*MCSE*	*SD*	*MCSE /SD in %*
VAagr	0.0010	0.0590	0.98	0.0000	0.0000	2.87	0.0000	0.0000	2.20
VAind	0.0020	0.1690	0.98	0.0000	0.0000	2.25	0.0000	0.0000	2.22
VAser	0.0100	0.4670	2.10	0.0000	0.0000	2.61	0.0000	0.0000	1.76
GDPcap	0.0120	0.5400	2.29	0.0000	0.0000	2.07	0.0000	0.0000	2.14
POPdens	0.0060	0.2820	2.25	0.0000	0.0001	2.01	0.0000	0.0001	2.10
ENVexp	0.0010	0.0630	1.69	0.0001	0.0007	6.47	0.0001	0.0008	7.28
ENVinc	0.0020	0.0800	2.02	0.0000	0.0000	4.41	0.0000	0.0000	3.09
ENGOmem	0.0010	0.0510	1.89	0.0000	0.0000	4.66	0.0000	0.0000	7.39

MCSE = Monte Carlo Standard Error, SD = Standard Deviation; *Source*: authors' calculations.

Part IV

Policy Design Studies

Integrating Ecological Indicators into Federal-State Fiscal Relations

A Policy Design Study for Germany

This article has been published as

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Abstract: Protected areas (PA) provide conservation benefits and ecosystem services that spill over the boundaries of jurisdictions to other regions. In this paper we analyse the foundations of and design options for ecological fiscal transfers (EFT) that may internalize such positive external effects. We propose a model for integrating ecological indicators into the intergovernmental fiscal transfer system between federal and state-level governments in Germany. Our approach is performance-oriented and would thus compensate those states that designate an above-average share of their area for nature conservation purposes. The suggested EFT design builds upon the existing fiscal equalization system and complies with the legal requirements for indicators determining fiscal needs. We employ an econometric analysis to demonstrate that, on average, sparsely populated states in Germany provide more PA per capita and would thus be eligible for increased fiscal transfers. A quantitative model of the fiscal transfer scheme is then used to estimate the marginal financial effects of integrating ecological indicators into federal-state fiscal relations in Germany. Moving beyond the specific case presented, we discuss the implications in terms of the specific role of EFT as a policy instrument within the broader conservation policy mix.

Keywords: ecological fiscal transfers, fiscal federalism, interjurisdictional spillover effects, multi-level governance, protected areas, Germany

JEL codes: H77, Q28, Q57, R14

5.1 Introduction

The unprecedented scale of biodiversity loss and ecosystem degradation has increasingly come to light (MEA, 2005). In many respects, this can be considered a problem of undersupplied public goods and services (Perrings and Gadgil, 2003). The benefits of biodiversity and ecosystem conservation have not yet been sufficiently integrated into decision making (Daily et al., 2009; TEEB, 2011). As a result of such insights, the role, functioning and interplay of conservation policy instruments have started to attract greater attention from concerned individuals and institutions in society, politics and academia (Larigauderie and Mooney, 2010; Nesshöver et al., 2016; Ring and Barton, 2015). Within the conservation policy mix, different policy instruments address different groups of actors: payments for environmental services (PES) address private land users, while ecological fiscal transfers (EFT) address public actors in their role as providers of environmental public goods (Ring and Schröter-Schlaack, 2015).

EFT close an important gap in the policy mix by internalizing conservation costs and benefits within the decision-making rationale of public actors. While the designation of protected areas (PA) builds on nature conservation laws, i.e. regulatory instruments, economic instruments such as EFT to local and state governments modify fiscal transfer schemes by considering PA as an additional indicator for distributing public money across governmental levels. In this way EFT change the nature of the incentives inherent in fiscal transfer schemes and help to create among public actors a mind-set more favourable to biodiversity conservation (Santos et al., 2015). EFT in Brazil and Portugal compensate decentralized governments for management and/or opportunity costs entailed by hosting PAs. Thus, EFT acknowledge fiscal needs for existing PAs and provide incentives to designate additional PAs (Droste et al., 2016, 2017c; Grieg-Gran, 2000; Loureiro, 2002; May et al., 2002; Ring, 2008c) and may improve the management of existing PAs (Loureiro, Pinto, and Motta, 2008). This is particularly important in view of the severe biodiversity conservation funding shortfalls in relation to politically set targets (McCarthy et al., 2012). Scaling up the finance mechanism for biodiversity is gaining increasing momentum and EFT schemes have more recently been considered as one of the necessary ingredients of environmental fiscal reforms around the globe (OECD, 2013). Internalizing intergovernmental spillover benefits from conservation at local levels (Ring, 2008a) may help reach the above-mentioned political standards for biodiversity conservation, since external effects are – at least partly – reduced (see Baumol and Oates, 1971, for a similar argument regarding the internalization of environmental damage costs).

Proposals for EFTs have been put forward for Switzerland (Köllner, Schelske, and Seidl, 2002), Poland (Schröter-Schlaack et al., 2014), Indonesia (Irawan, Tacconi, and Ring, 2014; Mumbunan, 2011), India (Kumar and Managi, 2009) and the state of Saxony in Germany (Ring, 2008b). Moreover, Farley et al. (2010) discuss the possibility of upscaling the transfer mechanism to the global level (as so-called International Payments for Ecosystem Services). In practice, only 'national or state to municipal level' EFT have been implemented. In federalist countries such as Brazil and Germany, or countries with more than two government levels, intergovernmental fiscal transfers exist between the federal (i.e. national) and the state (or regional) level, providing the states (or regions) with financial resources to fulfil their respective public functions. Very often, such state governments play an important role in nature conservation and the designation and/or management of protected areas. It therefore follows that EFT also need to be considered in federal-state (or nation-region) fiscal relations. So far, few concrete proposals for incorporating EFT policy into federal-state fiscal relations have been put forward, for example, for Brazil. Building on the so-called FPE Verde, Cassola (2011, 2014) has modeled and presented EFT policy options that integrate PA-related indicators into the State Participation Fund (Fundo de Participação dos Estados – FPE), a major fund for tax revenue distribution between the federal and the state level in Brazil.

Against this background, we seek to elaborate on the possibility of integrating ecological indicators into federal-state fiscal relations and use the German fiscal transfer system as an example. Our analysis proceeds in the following way: after elucidating the general rationale of (ecological) fiscal transfers in the following section, we implement a three-step approach to policy analysis that was developed especially to take account of the institutional embedding of policy instruments as well as their interplay (Ring and Schröter-Schlaack, 2015, pp. 148 ff.). The *first step* is to identify the institutional context (third section). Here, we elaborate on both the German institutional context of nature conservation, in particular the importance of state governments for nature conservation, and the (potential) role of (ecological) fiscal transfers. The *second step* is to identify knowledge gaps and choose methods for analysing them (fourth section): we develop empirical arguments to justify the integration of conservation-related indicators into the German fiscal transfer system at federal level. The *third step* is to evaluate policy instrument design options (fifth section). Here, we employ a quantitative benchmark factor model based on different PA-related ecological indicators (see Schröter-Schlaack et al., 2013). In the sixth section, we broaden the scope beyond these case specifics, discussing EFT design options and their implications, which are also of general relevance for other institutional contexts. The seventh section concludes with a brief reflection of lessons learnt for federal–state fiscal relations regarding EFT.

5.2 Rationales for Fiscal Transfers and the Integration of Ecological Indicators

In line with the first step of the policy analysis, we begin by analysing the institutional context. Germany is a federalist state comprising 16 states including three city states (Berlin, Hamburg and Bremen), the so-called Länder (Preamble and Art. 20 I German Constitution, also called Basic Law). According to Article 72 et seq. of the Basic Law the federal level of government has comprehensive legislative powers by which it can create a unified legal framework in many fields of law. The Länder are responsible for the execution and implementation of federal laws (Art. 83 Basic Law). This holds true for the designation and management of most PA categories in Germany, including Natura 2000 sites. The Länder thus have a key role in financing and implementing nature conservation, as they need to provide the necessary administrative capacity and funding to (at least partially) endow support programmes for private landholders. Annual costs for implementing and managing the Natura 2000 network alone have been estimated to be around €620 million for Germany (Gantioler et al., 2010). Fiscal transfers are an important source of income for the Länder, as they provide up to 28% of the total state budget per capita (see Table 5.3 later). An uneven distribution of PA (and hence an unequal distribution of conservation costs) would therefore justify compensating those states that provide above-average PA within the fiscal transfer scheme.

The German system of fiscal transfers between the federation and the 16 states (Länder) redistributes tax revenue both vertically (i.e. between federal and state level) and horizontally (i.e. balancing unequal fiscal capacities among different states) (BMF - Federal Ministry of Finance, 2015). Its aim is to enable the administrative authorities to fulfil their public functions in order to 'ensure uniform living standards' throughout the country (Basic Law, Art. 106; see also BMF, BMF - Federal Ministry of Finance, (2015)).[37] There are several stages of tax revenue distribution, including one of horizontal financial equalization between the German Länder, where poor states receive adjustment payments funded by the wealthier states to match fiscal capacity (i.e. mainly the states' tax income) with fiscal needs (BMF - Federal Ministry of Finance, 2015). As per capita fiscal needs are assumed to be the same among all the states, population numbers serve as the main indicator to calculate fiscal needs. There are, however, two important modifications in place: for both the densely populated city states and the three sparsely populated states, population numbers are increased calculatorily to account for population density-dependent above-average fiscal needs. Hence, the horizontal fiscal transfers are modified according to a U-shaped function in order to ensure there is sufficient fiscal capacity per capita. As a consequence, both the densely populated city states Berlin (BE), Hamburg (HH) and Bremen (HB) and the most sparsely populated states of Mecklenburg-Western Pomerania (MV), Brandenburg (BB) and Saxony-Anhalt (ST) are ascribed a calculatory increase in their actual population,[38] the so-called Einwohnerveredelung (Lenk, 2004). The Standards Act (MaßstG, 2009, §8) defines that the above-average needs of these states have to be determined by objective indicators showing an abstract higher need. That is to say, it cannot be public spending per se that determines higher fiscal need, not least because higher spending might be determined rather by higher fiscal capacity than by higher fiscal need.

Several public finance studies have analysed the relationship between fiscal needs and fiscal capacity in the context of the German federal system structure in order to demonstrate that the assumed above-average fiscal needs per capita are indeed an empirical pattern. By comparing city states[39] with similarly large cities that have surrounding areas under their administration, Hummel and Leibfritz (1987) show that city states are entitled to receive compensation because they provide public goods with positive spillover effects to the states surrounding them (Hummel and Leibfritz, 1987).

[37] The legal basis for implementation is the Financial Equalisation Act (Finanzausgleichsgesetz – FAG) and the Standards Act (Maßstäbegesetz – MaßstG).

[38] A factor of 1.35 for the city states of Bremen (HB), Hamburg (HH) and Berlin (BE) and, for the sparsely populated states, 1.05 for Mecklenburg-Western Pomerania (MV), 1.03 for Brandenburg (BB) and 1.02 for Saxony-Anhalt (ST). See Appendix A.2 for a formal description of the equalization scheme.

[39] City states are a peculiar characteristic of the German federal system. Bremen, Hamburg and Berlin are states that consist solely of the cities' territory, with no surrounding administrative areas.

Such above-average fiscal needs have further been substantiated by Eltges et al. (2001), who find above-average fiscal needs in city states due to social services provision and higher unemployment and crime rates. Additionally, they demonstrate slightly above-average fiscal needs per capita in sparsely populated states that provide road infrastructure and execute public responsibilities related to agriculture and forestry, among other things. Similar findings with regard to sparsely populated states have been presented by Seitz (2002), who finds a negative correlation between per capita infrastructure requirements and population density due to the lack of returns to scale from industrial or service agglomerations. Seitz therefore concludes that there is a substantiated above-average fiscal need per capita in sparsely populated states and that this fact should be accounted for within the fiscal equalization scheme.

All these studies have provided objective indications, backed up by empirical evidence, that above-average fiscal needs in both sparsely and densely populated states are a structural condition within the German federation. Legal judgements related to the issue acknowledge, furthermore, that calculatory modifications of inhabitant numbers to reflect above-average fiscal needs are in accordance with German Basic Law and its principle of solidarity (BVerfGE, 1986, 1992, 1999). Based on this, we now proceed with the second step of our policy analysis: in analogy to previous studies, we apply an econometric analysis of the relation between PA distribution and public spending for nature conservation among German states in order to demonstrate whether the coverage and category of PA also constitute a structural condition eligible for recognition in the German fiscal equalization system.

5.3 Empirical Approach: The Distribution of Protected Areas and Spending on Nature Conservation among German States

Although a systematic integration of environmental considerations into intergovernmental fiscal schemes has already been proposed for Germany (Czybulka and Luttmann, 2005; Möckel, 2013; Perner and Thöne, 2007; Ring, 2002, 2008b; Sachverständigenrat für Umweltfragen (SRU), 1996, 2002; Schröter-Schlaack et al., 2013) there is only limited empirical information available to date about fiscal needs for ecological public functions and their financial consideration within intergovernmental fiscal relations. Especially regarding the integration of conservation-related ecological public functions, there is not yet conclusive evidence for an objective indicator of above-average fiscal need per capita for federal states with above-average PA. Focussing on aspects of the states' legal obligations and competencies for conservation, Czybulka and Luttmann (2005) argue that there are substantiated reasons to assume an above-average fiscal need for conservation and the provision of related ecological public functions in sparsely populated states, but they do not provide quantitative evidence for this claim. Seitz (2001) provides quantitative evidence that European Natura 2000 sites are not strongly correlated with population density, but does not consider other (i.e.

national and regional) PA categories. Our contribution to the literature is to provide an empirical analysis of the spatial distribution of PA in German states: is there a significant correlation between population density and PA coverage in Germany, considering all PA categories? Since fiscal need is calculated on a per capita basis, a significant, negative correlation would provide (i) evidence of above-average fiscal needs per capita relative to the provision of conservation-related public goods and (ii) a justification for modifying the German federal financial equalization system by considering conservation-related indicators, as has been suggested previously (Czybulka and Luttmann, 2005; Möckel, 2013; Schröter-Schlaack et al., 2013).

5.3.1 Data

The Leibniz Institute of Ecological Urban and Regional Development (IOER) monitors data on spatial development such as PA coverage at state level (IOER, 2015; Walz and Schumacher, 2010). The so-called IOER Monitor includes two terrestrial PA categories relating to landscape and nature protection: (1) 'nature and species conservation', referring to the stricter German PA categories of national park, nature reserve and Natura 2000 site as well as the core areas of biosphere reserves, and (2) 'landscape protection', referring to nature parks and landscape reserves as well as buffer zones and transition areas in biosphere reserves with fewer land-use restrictions. Spatial overlaps are dealt with by taking only the PA category with stricter land-use restrictions into account. PA data is measured biannually. The IOER Monitor also provides data on population density. The IOER data set does not include any marine PA. The Federal Statistical Office provides data on the states' GDP per capita, value added per sector, and net public spending on environmental protection and nature conservation (Statistisches Bundesamt, 2015, and personal communication).

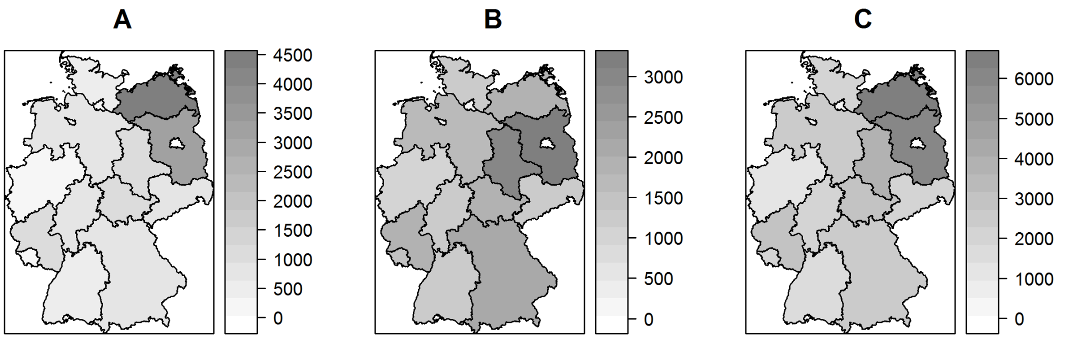

FIGURE 5.1: Spatial distribution of PA indicators for 2010 in Germany. (A) nature and species conservation, (B) landscape protection, (C) total PA. Units: m^2 per capita. *Source*: authors' elaboration based on IOER (2015).

5.3.2 Graphical Illustration

Figure 5.1 maps the distribution of three different PA indicators ('nature and species conservation', 'landscape protection' and 'total protected area') per capita for 2010 across the German Länder. As can be seen, the variation of 'nature and species conservation' is stronger than that of 'landscape protection', which is to say the latter is more equally distributed. Furthermore, there is a clear pattern that the least populated north-eastern region (i.e. the states of Mecklenburg-Western Pomerania and Brandenburg) provides most 'nature and species conservation' area as well as total PA per capita.

Figure 5.2 relates 2010 PA per capita to per capita public expenditure on nature conservation and environmental protection for the German Länder. It shows an unequally distributed share of PA between the Länder on the one hand (for strictly protected PA categories and for total PA) and public spending on the other. It illustrates that there is an exponentially declining relation between PA per capita and population density on the one hand and a more or less U-shaped relation between net public environmental and conservation expenditure per capita and population density on the other (see the second degree polynomial trend line). This illustrates our argument graphically: sparsely populated states provide a public good (with positive spillover effects) and have higher expenditures in the environmental and conservation sector – while densely populated states have high per capita environmental expenditure but do not provide much conservation. Therefore, we see PA as a suitable and objective indicator for above-average fiscal needs. However, this does not yet constitute an empirical proof of a significant correlation between PA per capita and population density. To this end, we next employ an econometric panel data analysis to test the null hypothesis that there is no significant correlation between PA and population density (Seitz, 2001).

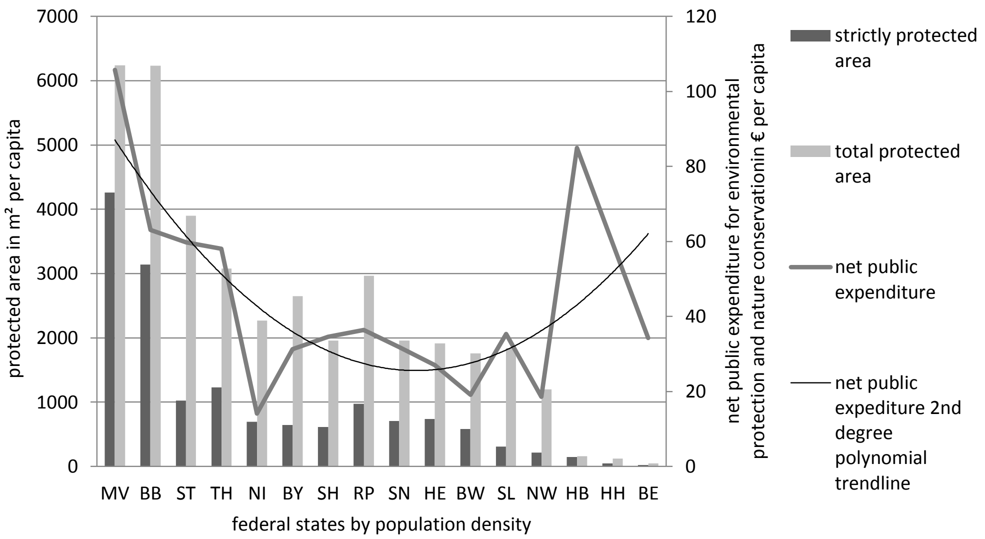

FIGURE 5.2: PA and respective public expenditure per capita for 2010 in Germany. *Source*: IOER (2015) and Federal Statistical Office (personal communication), figure adapted from Droste (2013)[40]

5.3.3 Econometric Model

To estimate the relation between variables, regressions are computed in the **R** environment (R Development Core Team, 2013). The ***plm*** package (Croissant and Millo, 2008) is employed to deal with unobserved heterogeneity among German Länder using individual specific fixed effects regressions. The dependent variable is protected area in square metres per capita, $PA.cap$ (using different PA categories such as $nat.cap$ for nature and species conservation, $land.cap$ for landscape protection or $tot.cap$ for total PA). Independent variables are GPD per capita ($GDP.cap$), the share of value added by agriculture (agr) and industry (ind) as a percentage of total value added by these two sectors, and public expenditure on environmental protection and nature conservation per capita in constant €2005 prices ($spend.cap$). A log–log transformation is employed. Furthermore, an integer year variable is used to detrend the data. This gives the general model structure:

$$\begin{aligned}\ln(\text{PA.cap}_{it}) = \beta_1 \ln(\text{pop.dens}_{it}) + \beta_2 \ln(\text{GDP.cap}_{it}) + \beta_3 \ln(\text{agr}_{it}) + \\ \beta_4 \ln(\text{ind}_{it}) + \beta_5 \ln(\text{spend.cap}_{it}) + \beta_6(\text{year}) + \mu_i + \epsilon_{it}\end{aligned} \tag{5.1}$$

with $i = 1, ..., 16$ entities (Länder), $t = 2006, 2008, 2010$ as the time index, the individual error component μ_i and an idiosyncratic error term ϵ_{it}, which is assumed to be normally distributed around mean zero and to be independent from regressors. Standard errors are computed with robust covariance matrix estimators à la Driscoll and Kraay (1998) with a maximum lag window of $m(T) = 1$ and estimation type $HC3$ (Millo, 2014; Zeileis, 2004) to account for heteroscedasticity, serial and spatial correlation.[41]

5.3.4 Results

Regressions for different dependent variables ('nature and species conservation', 'landscape protection' and 'total PA') are reported in Table 5.1. Summary statistics can be found in the appendix A.1. Controlling for socio-economic variables we find a significant, negative correlation between the logarithm of total PA in square metres per capita and population density (Model 6). This correlation provides sufficient evidence for a structural condition of the federation, namely, sparsely populated states providing more PA per capita. We also find a significant, negative correlation between GDP per capita and PA per capita (i.e. for nature and species conservation and total PA). This indicates that on average more PA are designated in relatively poorer states. For total PA, the value added by the agricultural and industry sectors is positively correlated.

[40] German Länder: MV: Mecklenburg-Western Pomerania, BB: Brandenburg, ST: Saxony-Anhalt, TH: Thuringia, NI: Lower Saxony, BY: Bavaria, SH: Schleswig-Holstein, RP: Rhineland-Palatinate, SN: Saxony, HE: Hesse, BW: Baden-Württemberg, SL: Saarland, NW: North Rhine-Westphalia, HB: Bremen, HH: Hamburg, BE: Berlin.

[41] The data, R code and files used for analysis and plotting can be found at: `https://github.com/NilsDroste/EFT-DE`.

While public expenditure for environmental protection and nature conservation is significantly and positively correlated with nature and species conservation per capita, it is significant and negative for landscape protection and total PA – which may indicate that it is a poor indicator for conservation performance due to its composite aggregate of both environmental protection and conservation expenditure. The adjusted R^2 indicates that Models 1 to 4 have a really poor fit, taking the variable-to-sample size ratio into account. Looking at the overall picture – taking all PA categories into account – the adjusted R^2 values suggest that Model 5 is preferable to Model 6.

TABLE 5.1: State-level regressions

	Dependent variable					
	ln(nat.cap)		ln(land.cap)		ln(tot.cap)	
	(1)	(2)	(3)	(4)	(5)	(6)
ln(pop.dens)	-0.292^{***}	0.395	-1.095^{***}	-1.211^{**}	-1.162^{***}	-0.925^{***}
	(0.075)	(0.601)	(0.378)	(0.581)	(0.316)	(0.198)
ln(GDP.cap)	-0.995^{***}	-0.889^{***}	-0.296^{*}	-0.365^{**}	-0.651^{***}	-0.644^{***}
	(0.258)	(0.192)	(0.146)	(0.171)	(0.066)	(0.038)
ln(VA.agr)		-0.032		0.097^{***}		0.043^{***}
		(0.019)		(0.021)		(0.013)
ln(VA.ind)		0.701		0.058		0.349^{*}
		(0.543)		(0.153)		(0.197)
ln(spend.cap)		0.040^{**}		-0.075^{***}		-0.027^{***}
		(0.017)		(0.023)		(0.010)
year	0.025^{***}	0.032^{***}	0.010	0.003	0.015^{***}	0.014^{***}
	(0.002)	(0.004)	(0.006)	(0.006)	(0.005)	(0.002)
Observations	48	48	48	48	48	48
R^2	0.356	0.404	0.198	0.297	0.561	0.604
Adjusted R^2	-0.043	-0.077	-0.300	-0.271	0.288	0.283
F Statistic	5.354^{***}	2.937^{**}	2.380^{*}	1.832	12.335^{***}	6.598^{***}

The panel data sample is balanced with $n = 16, T = 3, N = nT = 48$. Robust standard errors are reported in parentheses below the estimated coefficients. Individual coefficients are indicated by a $^{*}10\%$, $^{**}5\%$, or $^{***}1\%$ significance level. The models use an individual fixed effects specification.

Previous reforms of the fiscal transfer system in Germany have acknowledged the fact that both densely and sparsely populated states have additional fiscal needs when it comes to fulfilling their public functions (Eltges, Zarth, and Jakubowski, 2001; Hummel and Leibfritz, 1987; Seitz, 2002). Regarding nature conservation, there have been legal arguments in favour of EFT in Germany (Czybulka and Luttmann, 2005) but no empirical analysis beyond Natura 2000 sites (Seitz, 2001). The overall negative correlation of total PA per capita with population density (Model 5) provides evidence for a structural condition within the federation, namely, that sparsely populated states on average provide more PA per capita, most likely due to a higher propensity to designate PA in regions with great natural endowments. Considering that nature conservation efforts impose management costs, we have thus provided evidence that PA per capita is an objective indicator of an above-average fiscal need for conservation in sparsely populated states in Germany, and we now proceed by proposing policy options for including suitable indicators in the fiscal transfer system in Germany.

5.4 Assessment of Policy Options: Modelling Ecological Fiscal Transfers in Germany

Having established that there is indeed a structural condition for the distribution of PA in Germany, namely, higher provision where there is lower population density, we proceed to step three in our policy analysis by evaluating the fiscal effects of different EFT design options.[42]

5.4.1 Ecological Fiscal Transfer Model

In the existing German fiscal equalization scheme, tax revenue is transferred from wealthier to poorer states. In this way, differences in fiscal budget per capita among federal states are substantially equalized (Lenk, Glinka, and Sunder, 2015; Lenk and Kuntze, 2012). The respective formulae can be found in appendix A.2.

Although different options are discussed elsewhere (see Schröter-Schlaack et al., 2013), we restrict the analysis here to a relatively simple approach of accounting for above-average fiscal needs per capita. Building on the modification of population numbers for sparsely and densely populated states, we suggest increasing the population numbers by a factor eco_i to account for conservation-related above-avarage fiscal needs of states. Equation 5.2 gives the ecological benchmark assessment for the conservation factor eco_i .

$$eco_i = 1 + f_{eco}(\frac{PA_i}{PA_{DE}} - 1) \tag{5.2}$$

where PA_i is the PA per capita in state i. The benchmarking consists in a ratio of state provision of PA and federal average of PA per capita PA_{DE} minus 1. The benchmark factor eco_i will be f_{eco} times larger than 1 if PA coverage in state i is above average and f_{eco} times smaller than 1 if it is below average.[43] To account for the state's relative conservation performance, we suggest integrating the factor eco_i into the fiscal need formula in analogy to previous decisions to take account of above-average fiscal needs for sparsely populated states (see Appendix 5.6, Equation 5.4). In order to provide various policy options for a political process that is ultimately based upon negotiations between he German central government and the Länder, we show results for three different benchmark conservation factors eco_i. The first, $eco_{NCA,i}$ is formed using 'nature and species conservation' area per capita. The second is a weighted per

[42] For more detail and further design options, see Schröter-Schlaack et al. (2013) and Droste (2013).

[43] Factor f_{eco} is a weighting factor that reflects the extent to which differences in PA coverage are taken into account. It is set to 0.1 to yield reasonable marginal fiscal transfer changes (Schröter-Schlaack et al., 2013).

capita sum of 'nature and species conservation' (weight = 0.8) and 'landscape protection' $eco_{LPA,i}$ (weight = 0.2), since there are different conservation benefits provided by different PA categories and the spatial distribution of both categories differs among states (see also later for a discussion of the issue). The third indicator $eco_{TPA,i}$ is based on an unweighted total PA per capita (see Table 5.2).

TABLE 5.2: Conservation factors by different nature conservation area categories for 2010

Federal State	Population	'Nature and species conservation' area (NCA) in km^2	NCA area in m^2 per capita	NCA conservation factor eco_{NCA}	'Landscape protection' area (LPA) in km^2	LPA area in m^2 per capita	LPA conservation factor eco_{LPA}	Weighted factor 0.8 eco_{NCA} + 0.2 eco_{LPA}	'Total Protected Area' (TPA) in km^2	TPA area in m^2 per capita	TPA conservation factor eco_{TPA}
BW	10,753,880	6,256.50	582.00	**0.98**	12,620.30	1,174.00	**0.98**	**0.98**	18,876.77	1,755.00	**0.98**
BY	12,538,696	8,042.70	641.00	**0.99**	25,115.80	2,003.00	**1.04**	**1.00**	33,158.49	2,644.00	**1.03**
BE	3,460,725	68.60	20.00	**0.90**	95.40	28.00	**0.90**	**0.90**	164.04	47.00	**0.90**
BB	2,503,273	7,812.70	3,121.00	**1.35**	7,724.30	3,086.00	**1.12**	**1.30**	15,536.99	6,207.00	**1.19**
HB	660,706	74.40	113.00	**0.92**	27.90	42.00	**0.90**	**0.91**	102.69	155.00	**0.91**
HH	1,786,448	80.00	45.00	**0.91**	138.90	78.00	**0.91**	**0.91**	219.00	123.00	**0.91**
HE	6,067,021	4,476.40	738.00	**1.01**	7,136.80	1,176.00	**0.98**	**1.00**	11,613.20	1,914.00	**0.99**
MV	1,642,327	6,979.90	4,250.00	**1.51**	3,246.50	1,977.00	**1.04**	**1.42**	10,249.53	6,241.00	**1.20**
NI	7,918,293	5,478.00	692.00	**1.00**	12,432.70	1,570.00	**1.01**	**1.00**	17,958.39	2,268.00	**1.01**
NW	17,845,154	3,817.90	214.00	**0.93**	17,589.40	986.00	**0.97**	**0.94**	21,407.27	1,200.00	**0.96**
RP	4,003,745	3,891.30	972.00	**1.04**	7,981.10	1,993.00	**1.04**	**1.04**	11,892.29	2,970.00	**1.04**
SL	1,017,567	315.90	310.00	**0.94**	1,559.20	1,532.00	**1.01**	**0.96**	1,875.12	1,843.00	**0.99**
SN	4,149,477	2,928.70	706.00	**1.00**	5,194.40	1,252.00	**0.99**	**1.00**	8,123.09	1,958.00	**0.99**
ST	2,335,006	2,392.50	1,025.00	**1.05**	6,707.20	2,872.00	**1.10**	**1.06**	9,099.74	3,897.00	**1.08**
SH	2,834,259	1,737.90	613.00	**0.99**	3,807.60	1,343.00	**1.00**	**0.99**	5,545.47	1,957.00	**0.99**
TH	2,235,025	2,749.30	1,230.00	**1.08**	4,140.10	1,852.00	**1.03**	**1.07**	6,889.45	3,082.00	**1.05**
all	81,751,602	57,102.80	698.00		115,517.60	1,413.00			172,711.50	2,113.00	

Source: authors' calculation based on IOER (2015), Schröter-Schlaack et al. (2013) and Droste (2013).

5.4.2 Marginal Fiscal Transfer Changes upon Integration of Ecological Indicators

The integration of ecological indicators changes fiscal transfers compared with the current distribution. We quantify the marginal changes in transfers for the three different conservation factor scenarios (see previous subsection). Table 5.3 gives marginal transfer changes as of 2010 for each of the Länder if three different ecological indicators were integrated into the current German financial equalization system, comparing them to the status quo. As can be seen in Table 5.3, tax revenue per capita is equalized through fiscal transfers among German states (status quo scenario). When ecological indicators are introduced they cause a deviation from the status quo fiscal transfers and we indicate with a minus where states suffer a loss compared with the status quo. It becomes clear that, regardless of the specific ecological indicator eco i chosen, winners and losers stay more or less the same over the three scenarios (except BY, NI and SN). However, the magnitude of transfers changes drastically in some cases (e.g. for MV and SL) across different indicators due to the different spatial distribution patterns of different PA categories.

TABLE 5.3: Marginal fiscal transfer changes as of 2010 for three different scenarios involving the integration of different PA-based ecological indicators into the German financial equalization system.

Scenarios	Status Quo			Nature and species conservation		Nature and species conservation (0.8) + landscape protection (0.2)		Total PA	
Federal State	Tax revenue **before** fiscal transfers in € per capita	Tax revenue **after** fiscal transfers in € per capita	Fiscal transfers in € per capita	EFT in € per capita	Relative change per capita transfers to status quo in %	EFT in € per capita	Relative change per capita transfers to status quo in %	EFT in € per capita	Relative change per capita transfers to status quo in %
BW	3,441	3,282	−159	−170	−6.7	−169	−6.4	−168	−5.7
BY	3,611	3,331	−280	−291	−3.7	−286	−1.9	−274	2.3
BE	2,835	3,942	1,107	1,056	−4.5	1,056	−4.5	1,056	−4.6
BB	2,777	3,007	230	408	77.4	385	67.1	329	43.2
HB	3,152	4,048	896	852	−4.8	851	−5.0	848	−5.3
HH	4,377	4,340	−37	−67	−79.2	−67	−78.8	−66	−77.8
HE	3,643	3,354	−289	−294	−1.7	−295	−2.0	−297	−2.8
MV	2,622	2,960	338	600	77.5	552	63.2	438	29.7
NI	3,018	3,067	49	48	−1.0	49	1.4	52	7.4
NW	3,077	3,103	26	2	−93.2	4	−85.9	8	−68.4
RP	2,953	3,055	102	122	19.1	122	19.3	123	19.9
SL	2,908	3,041	133	105	−21.0	112	−16.2	127	−4.9
SN	2,664	2,954	290	290	0.2	289	−0.3	286	−1.3
ST	2,663	2,961	298	322	8.0	328	10.0	342	14.5
SH	3,022	3,076	54	105	−21.0	112	−16.2	127	−4.9
TH	2,647	2,944	297	336	13.2	331	11.6	320	7.9

Source: authors' calculation based on Droste (2013) and Schröter-Schlaack et al. (2013), PA data for 2010 from IOER (2015).

5.5 Discussion: Federal-State Level EFT Design Options and their Implications

In many federalist states such as Germany, (regional) state governments are lacking in adequate financial resources for nature conservation, while often being responsible for the designation and management of PA. Furthermore, the existing incentive structure of fiscal transfers is not conducive to taking conservation benefits into account when deciding about allocating state budget among different public responsibilities. Thus, given tight (public) budgets in general and a severe lack of conservation financing more specifically (McCarthy et al., 2012), EFT constitute an innovative and complementary financing instrument in the conservation policy mix.

EFT schemes based on PA indicators that have been implemented to date usually involve general purpose transfers, meaning that these are not earmarked for spending on nature conservation. However, conservation-related indicators serve to bind the distribution of intergovernmental fiscal transfers to the existence of conservation efforts displayed in the ecological indicators applied. EFT thus create an incentive to conserve nature in order to access this part of the budget. The costs of providing conservation would be compensated (at least partly) by acknowledging PA indicators for fiscal transfers, thereby internalizing conservation spillover benefits. In this sense, EFT share some characteristics with PES as they incentivize decision makers to change their behaviour in an environmentally friendly way. Since neither implemented EFT schemes in Brazil or Portugal, nor our design proposal, are based on actual marginal costs and benefits, we do not claim that the internalization achieved is optimal in an economic sense. However, we argue that a (partial) internalization of PA spillovers would still increase incentives to comply with predefined political conservation standards, such as a certain share of PA on total state area (BNatschG, 2009). Rather than aiming at optimal solutions, standard-price approaches (Baumol and Oates, 1971) as well as evolutionary strategies in environmental policy (Ring, 1997) provide signals in the right direction.

Our proposed approach is based on an assessment of how much PA is actually provided by individual states compared with the average; as such, it is performance based (see Table 5.2). It requires no additional budget from the (national) federal government but creates conservation incentives by greening the indicators for tax revenue allocation (Droste et al., 2017c). Hence, there is no increase in the overall amount of money available, and some states will receive less with EFT than under the status quo (see Table 5.3). This is due to the fact that these states underperform or are below average in relevant nature conservation activities. While this may be seen as a dynamic incentive for conservation, which introduces elements of competitive federalism by virtue of its performance-based design (Oates and Schwab, 1988), the annual amount of fiscal transfers (and thus a share of state total budget) would depend on yearly conservation performance in terms of PA designated by the jurisdiction compared with other jurisdictions. That is to say, the incentive element of EFT alone cannot ensure that there is sufficient conservation financing available, but it can act as a complement to general conservation financing by providing a fiscal incentive for public administrations to perform well in terms of conservation benchmarks.

Based on the foregoing remarks, a critical aspect is the choice of indicator. In Germany, the 16 states' fiscal needs are calculated on the basis of weighted population numbers, the weightings being derived from abstract and objective indicators for above-average fiscal needs. We have therefore developed an approach tailored to the

German system that includes an additional population weighting for providing conservation (see Table 5.2). Different distributional effects occur depending on the different indicators we have used to compute EFT. As can be seen from Table 5.3, there are substantial differences in transfers to individual states depending on the type of indicator chosen. Regarding the choice of indicators, we argue that stricter PAs very likely provide greater benefits for biodiversity conservation and hence greater interjurisdictional spillover benefits. Nevertheless, landscape protection also provides spillover benefits in terms of recreational and amenity services. Thus, our proposed combined and weighted indicator for EFT takes these factors into account (see Table 5.2). How different PA categories perform in terms of biodiversity conservation and ecosystem service provision and what this would imply for designing EFT indicators is, however, a question for future research. While we cannot provide a generally applicable solution in this paper, it becomes clear that there is room for manoeuvre in terms of political negotiations to counterbalance unacceptable burdens for individual states.

5.6 Conclusion

EFT close an important gap in the conservation policy mix. They explicitly address decentralized public actors such as state or municipal governments. Whereas there is a range of economic instruments directed towards private actors (such as tax reliefs, agri-environmental schemes or PES), there is no such variety aimed at public actors. Therefore, EFT provide a suitable instrument to address local, regional and state governments. PAs provide conservation benefits that spill over the boundaries of the jurisdictions, providing them to other regions (ten Brink et al., 2013). We have analysed the theoretical and empirical underpinnings of fiscal transfers and the rationales for including ecological indicators, and have presented design options for EFT in Germany that may internalize such positive external effects.

However, EFT cannot simply be transferred from one country to another. They need to be tailored according to the legal and institutional framework in place. This requires analysis of the institutional context, closing of knowledge gaps and derivation of an appropriate policy design from there (cf. Ring and Schröter-Schlaack, 2015, for the underlying policy analysis approach). Previous reforms of the German financial equalization system from the federal to the state level have been based on above-average fiscal needs in both densely and sparsely populated states and have led to a calculatory population increase for these states. We have shown econometrically that the same structural condition holds for the distribution of PA. There is a significant negative correlation between PA coverage per capita and population density across the German Länder. This provides a structural argument for an integration of ecological indicators into the current fiscal transfer system. We have presented a potential performance-oriented model that assesses the designation of different PA categories using the national average as a benchmark. States with above-average PA coverage per capita would be entitled to receive increased fiscal transfers, whereas states below the average would lose out. Such an EFT scheme transforms above-average PA coverage into a source of state revenue and builds closely on the legal and institutional setting of intergovernmental fiscal relations in Germany. The idea of performance-oriented EFT may well, however, be transferred to other states or even supra-national bodies (cf. Droste et al., 2016).

Looking beyond our particular policy design study, existing EFT schemes with PA-related indicators all focus on EFT to the local government level, regardless of whether the country is organized centrally (Portugal) (Santos et al., 2012) or federally (Brazil) (Grieg-Gran, 2000; Ring, 2008c). Our proposal may provide useful insights for other federal systems where the financial constitution regulates fiscal relations between the federal and the state level. In fact, federal to state-level EFT make it possible to take the interstate spillover effects of nature conservation into account. This promises to be especially relevant to large federalist countries with heterogeneous natural endowments such as Brazil, which is a major hotspot of global biodiversity and yet has a noticeably unequal spatial distribution in relation to biomes, PA, population and socio-economic characteristics (Cassola, 2011; Droste et al., 2017c). While initial policy proposals for federal-to-state EFT schemes in Brazil (Cassola, 2011, 2014), Switzerland (Köllner, Schelske, and Seidl, 2002) and India (Kumar and Managi, 2009) have been put forward, our approach is the first to consider the integration of indicators on conservation performance at state level into fiscal equalization between states. This provides a complementary design option that could be adapted elsewhere.

Acknowledgments

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Appendix

A.1 Summary statistics

TABLE 5.4: Summary statistics

Statistic	N	Mean	SD	Min	Max
Nature and species conservation area per capita in m^2 (nat.cap)	48	915.8	1,046.0	19.7	4,266.1
Landscape protection area per capita in m^2 (land.cap)	48	1,402.1	856.1	24.5	3,101.1
Total protected area per capita in m^2 (tot.cap)	48	2,317.9	1,737.4	44.6	6,256.8
Population density in persons/km^2 (pop.dens)	48	670.8	1,033.1	71	3,884
GDP in € per capita (GDP.cap)	48	28,452.4	7,842.7	19,610.3	50,691.0
Valued added agriculture as a percentage of total value added (VA.agr)	48	1.1	0.9	0.01	3.7
Valued added industry as a percentage of total value added (VA.ind)	48	28.6	6.5	15.9	39.0
Public expenditure environmental protection and nature conservation in €per capita (spend.cap)	48	41.5	22.7	10.0	100.4

Source: authors' calculations based on IOER (2015) and Statistisches Bundesamt (2015, and personal communciation), monetary values are in constant €2005 prices.

A.2 Financial Equalization Act

According to the Financial Equalization Act (FAG), fiscal capacity and fiscal need are defined as given in Equations 5.3 and 5.4. Adjustment payments result from comparing the fiscal capacity index FC_i and the equalization index FE_i of a state. If state i's FC is larger than its FE, the state pays transfers, and vice versa.

Fiscal Capacity Index (FC):

$$FC_i = S_i + \sum_{j=1}^{m} 0.64 M_{ij} \tag{5.3}$$

FC of state i is determined by the sum of state-level tax revenue S of state i and 64 per cent of the municipal-level tax revenue M of all municipalities j in state i.

Fiscal Equalization Index (FE):

$$FE_i = \frac{\sum_{k=1}^{n} S_k}{\sum_{k=1}^{n} g_1 P_k} + \frac{\sum_{k=1}^{n}\sum_{j=1}^{m} 0.64 M_{kj}}{\sum_{k=1}^{n} g_2 P_k} g_2 P_i \tag{5.4}$$

In principle, the German system assumes that the fiscal need per inhabitant is the same for all states. Therefore, the FE of state i is determined by the average tax revenue per capita at state level S among all $k = 1, ..., 16$ states multiplied by the weighted population P of state k plus 64 per cent of the average municipal tax revenue M of municipalities j of state k multiplied by the weighted population P of state i.[44] The fiscal transfers are then determined by a linear-progressive equalization function (FAG, 2013, §10) depending on the extent to which the relevant states diverge from the average. As can be seen from Equations 5.3 and 5.4, only 64

[44] The weight g_1 is 1.35 for the city states Bremen, Hamburg and Berlin and 1 for all other states. Weight g_2 is again 1.35 for the city states while a factor of 1.05 applies to Mecklenburg-Western Pomerania, of 1.03 to Brandenburg, and of 1.02 to Saxony-Anhalt. This means that fiscal need is basically the same for all states with a factorial increment of the population of the three city states and the three most sparsely populated states. The factor compensating sparsely populated states for above average fiscal needs is applied only at the municipal level.

per cent of the local authorities' tax revenues are taken into account in determining the states' fiscal capacity. Since local authorities have relevant fiscal needs and capacities and their public functions differ between the states, Lenk et al. (2015) call for municipal tax revenues to be acknowledged fully in the financial equalization. However, our EFT model is based on the existing formulae.

The ecological benchmark factor eco_i (see Equation 5.2) would be integrated on the municipal level by replacing g_2 by $g_i = g_2 eco_i$, where above-average fiscal needs for sparsely populated states have also been integrated.

A.3 An additional set of regressions

The following regressions have not been published with the original article. They are an addition that take better into account the supposedly U-shaped relation of PA per capita and environmental and conservation spending (see Figure 5.2). Again, as in the original regressions, Models 3-4, that is the ones on 'landscape protection' have a bad fit in terms of adjusted R^2. The ones on 'nature and species conservation' have a better fit than the original models but still a relatively poor one (Models 1-2). The adjusted R^2 values for the regressions on total PA per capita increase compared to the models originally reported (Table 5.1, Models 5-6). The general conclusion nevertheless holds – with an improved model fit. Population density and GDP per capita are significantly, negatively correlated with total PA per capita. Furthermore, the significant and negative correlation of environmental and conservation spending per capita ($spend.cap$), the significant and positive correlation of spending squared, and the increased adjusted R^2 in Models 5-6 indicate that a potentially U-shaped spending model specification has greater explanatory power compared to the monotonic log-log model specification orignally reported.

TABLE 5.5: State-level regressions including spend.cap squared

	Dependent variable					
	ln(nat.cap)		ln(land.cap)		ln(tot.cap)	
	(1)	(2)	(3)	(4)	(5)	(6)
ln(pop.dens)	−0.243	0.720	−1.316***	−1.426**	−1.222***	−0.815***
	(0.292)	(0.551)	(0.338)	(0.524)	(0.442)	(0.186)
spend.cap	−0.012***	−0.011***	0.005**	0.005**	−0.006**	−0.006*
	(0.002)	(0.002)	(0.002)	(0.003)	(0.003)	(0.003)
spend.cap^2	0.0001***	0.0001***	−0.0001***	−0.0001***	0.000 05**	0.000 04*
	(0.000 02)	(0.000 02)	(0.000 01)	(0.000 02)	(0.000 02)	(0.000 02)
ln(GDP.cap)		−0.606***		−0.544***		−0.538***
		(0.152)		(0.191)		(0.107)
ln(VA.agr)		0.006		0.065***		0.052***
		(0.021)		(0.006)		(0.015)
ln(VA.ind)		0.948		−0.118		0.422*
		(0.609)		(0.226)		(0.240)
year	0.028***	0.028***	0.014**	0.006	0.018***	0.013***
	(0.001)	(0.004)	(0.006)	(0.007)	(0.005)	(0.003)
Observations	48	48	48	48	48	48
R^2	0.439	0.552	0.356	0.395	0.550	0.649
Adjusted R^2	0.058	0.157	−0.080	−0.138	0.245	0.341
F Statistic	5.471***	4.397***	3.875**	2.330*	8.572***	6.611***

The panel data sample is balanced with $n = 16, T = 3, N = nT = 48$. Robust standard errors are reported in parentheses below the estimated coefficients. Individual coefficients are indicated by a $^*10\%$, $^{**}5\%$, or $^{***}1\%$ significance level. The models use an individual fixed effects specification.

Ecological Fiscal Transfers in Europe

Evidence-based design options of a transnational scheme

This article is based on a working paper and has been submitted to Ecological Economics and received a major revision decision. This is the revised version.

Droste, N., Ring, I., Santos, R., Kettunen, M (2016) Ecological Fiscal Transfers in Europe – evidence-based design options of a transnational scheme. *UFZ Discussion Paper 10/2016*.

Abstract: Ecological Fiscal Transfers (EFT) have recently gained attention as a promising instrument addressing public authorities to provide incentives for nature conservation. In parallel, both the EU and various European countries are exploring new mechanisms to mobilise funding to support biodiversity conservation. We develop a proposal for an EFT design within the supranational context of the EU and assess its potential effects with evidence-based estimates. We i) provide both a theoretical underpinning and a synthesis of the current EFT schemes and EU Nature Directives, ii) propose a model for EFT implementation within the existing EU funding mechanisms based on quantitative and qualitative conservation indicators, iii) analyse how resulting payments would be (spatially) distributed among European regions, and iv) discuss the model outcomes in terms of ecological effectiveness, distributive effects and cost-effectiveness. We thereby contribute to the debate about how to better integrate ecological public functions within multi-level and supra-national governance structures.

Keywords: Ecological Fiscal Transfers, European Union, Natura 2000 network, policy advice, spatial econometrics

JEL codes: C31, H77, H87, P48, R12, Q57

Highlights:

- a tailored proposal for upscaling ecological fiscal transfers to EU level
- empirical estimations of socio-economic and bio-geographical characteristics of beneficiaries
- evidence-based policy advice to improve effectiveness of conservation

6.1 Introduction – The Need for Innovation in Conservation Policies

While the Millennium Ecosystem Assessment (2005) and The Economics of Ecosystems and Biodiversity (TEEB) reports (2010) have successfully raised awareness about the importance of healthy ecosystems for human well-being, political measures have not yet been sufficient to halt the decline in biodiversity (Hooper et al., 2012; Waldron et al., 2013). Being at the international forefront in conservation efforts, the EU biodiversity strategy has indeed set ambitious goals for conservation but lacks implementation effectiveness (European Commission, 2011, 2015). The European Natura 2000 (*N2k*) network of protected areas (*PA*) is a cornerstone of the strategy since transnational habitat and species conservation networks play a crucial role in the protection of important natural heritage (Pereira and Navarro, 2015) and migratory species (Opermanis et al., 2012). However, while N2k provide substantial benefits to both biodiversity and people (ten Brink et al., 2013), the successful implementation yet lacks sufficient financing (Kettunen et al., 2011, 2017; Milieu, IEEP, and ICF, 2016; N2k Group, 2016).

In this context, there is an increasing interest in the supplementary use of economic instruments to both increase the financing for biodiversity conservation and improve the effectiveness and efficiency of conservation efforts (TEEB, 2010). For example, result-based wagri-environment measure are being increasingly used to improve the efficiency and behavioural changes of private land users (Burton and Schwarz, 2013; Matzdorf and Lorenz, 2010; Russi et al., 2016). Conservation policy through protected areas is primarily a public function (Ring, 2002). Hence, beyond instruments that address private land users (Vatn, 2015) the conservation policy mix is not complete without instruments that support public bodies in their function to conserve nature (cf. Ring and Barton, 2015).

An innovative instrument that addresses public bodies explicitly is Ecological Fiscal Transfers (EFT). EFT are an element of intergovernmental fiscal transfers that allocate tax revenue among different government levels according to ecological criteria such as the existence of protected areas (PA). EFT are promising in terms of conservation outcomes since i) they do not necessarily require additional funding as such but can be based on introducing changes to existing allocation schemes, and ii) they can be used to incentivise the creation of PA (Droste et al., 2017a,c; Grieg-Gran, 2000; May et al., 2002; Ring, 2008c; Santos et al., 2012). Originating from the Brazilian state of Paraná the instrument has spread among other Brazilian states (Droste et al., 2017c; Grieg-Gran, 2000; Loureiro, 2002; Loureiro, Pinto, and Motta, 2008; May et al., 2002; Ring, 2008c; Sauquet, Marchand, and Féres, 2014). As the first EU Member State, Portugal has introduced a fully fleshed EFT scheme from the national to the local governmental level for all PA categories in 2007 (Santos et al., 2012, 2015).[45] The idea of EFT has

[45] Since 2006, a small-scale EFT scheme exists in France which provides ecological transfers for municipalities in core zones of national parks or natural marine parks (Borie et al., 2014).

received international attention (May et al., 2002; Ring, 2008c) and it is gaining momentum regarding potential implementations in other states such as Switzerland (Köllner, Schelske, and Seidl, 2002), India (Kumar and Managi, 2009), Indonesia (Irawan, Tacconi, and Ring, 2014; Mumbunan, 2011), Germany (Ring, 2008b; Schröter-Schlaack et al., 2014), and France (Borie et al., 2014). Even an adaptation to the global level has been proposed (Farley et al., 2010).

Given the lack of finance for a successful implementation of the EU biodiversity conservation objectives, including the N2k network, and the potential of EFT to support conservation policies through financial incentives for conservation, the aim of this article is to explore a possible policy design for EU-wide implementation of an EFT scheme based on empirical evidence of the distribution of N2k areas and experience gained with existing EFT mechanisms. In order to provide the relevant background information, we introduce both a theoretical foundation for an EU-level EFT scheme, and synthesise current experience with EFT, N2k governance, and EU conservation financing (section 6.2). We then present a tailored proposal for a European EFT scheme (section 6.3). In a next step, we analyse the spatial distribution of simulated EFT payment flows among European regions within the proposed scheme (section 6.4). Finally, we discuss the potential outcomes of the proposed scheme in terms of conservation effectiveness, distributional effect and cost-effectiveness (section 6.5) and conclude with a note on the political economy of conservation (section 6.6).

6.2 Background – What Have We Learnt So Far?

In order to understand potential design options, we provide a brief theoretical underpinning for fiscal transfers in general and for introducing an EU-level EFT scheme in particular (section 6.2.1).We present the basic functioning of existing national and state-level schemes for local governments in Brazil and Portugal, where EFT were first implemented (section 6.2.2). For a suitable adaptation to the multi-level conservation governance structure of the EU, we elaborate on the implementation of N2k policies and their existing funding opportunities within the current EU (co-)financing schemes (section 6.2.3).

6.2.1 A Theoretical Foundation

In multi-level government structures the various levels each have their particular public functions which require corresponding public budgets. This is the main reason for revenue sharing and fiscal transfer schemes: to ensure sufficient finances for public functions at all government levels. Furthermore, there are often equity and efficiency considerations that determine the design of the fiscal system (Boadway and Shah, 2009). As a general guideline, the *principle of fiscal equivalence* (Olson, 1969) states that those jurisdictions who obtain the benefits of a policy should also bear the costs of delivering it. In the case of PA, where a decentral policy benefits other jurisdictions or serves higher level government interests, fiscal transfers may serve the internalisation of spill-over benefits between jurisdictions. By lowering the cost of provision they create incentives for an additional supply from respective government levels.

There are different forms of fiscal transfers (Boadway and Shah, 2009): *General-purpose transfers* supply sufficient funds for general public functions at the local or regional level. *Specific-purpose transfers* are designed to create incentives for lower-level government to provide specific public goods and services and are thus earmarked for particular spending objectives. The latter may be provided as matching grants that require co-financing from both higher and lower level government sources. A third and hybrid form are output-based or *performance-oriented transfers* which are conditional on the supply of a particular result but do not necessarily require that transfers received are spent on specified purposes.

In the context of *ecological public functions* (Ring, 2002), these design options have different implications for the financing of conservation policies. General-purpose transfers increase the general budget and, depending on how the receiving administration allocates the respective budget, may also increase conservation spending. Specific-purpose transfers are earmarked to support the implementation of a certain policy area. If – as in the case of PA – some benefits remain at local or regional level (e.g. amenity services and local water quality) and others spill over to the (inter-)national level (e.g. climate regulation and biodiversity conservation) (Gantioler et al., 2010; ten Brink et al., 2013), specific-purpose transfers in the form of matching grants are an option for internalisation. Performance-oriented transfers do not necessarily require that the obtained revenue is spent on a particular activity but require the supply of a specific result and thus maintain some decentral autonomy as to how the money is best spent and how the result is obtained. Through performance-based transfers the provision of a particular result becomes a source of income and greater supply is incentivised. Existing EFT schemes are both based on the logic of general-purpose transfers (they supply transfers based on the financing need for ecological public functions), and they transfer funds conditional on ecological indicators such as the (relative) coverage of PA (for details see section 6.2.2). Hence, in the context of fiscal terminology EFT can be considered performance oriented.

Within EU multi-level conservation governance structures there are thus arguments for different possible types of fiscal transfers or fund mechanism designs. From the perspective of EU-level interests, general-purpose transfers may not well serve the purpose of conservation policies since they lack a close tie to conservation spending or outputs. Specific-purpose transfers that are dedicated to particular programmes and activities serve two main and connected purposes: they earmark spending on conservation policies and could thus ensure that sufficient funding is available for conservation activities. Of these two, only the first is given, since a N2k funding gap remains and sufficient funding is not ensured (Kettunen et al., 2011, 2017; Milieu, IEEP, and ICF, 2016; N2k Group, 2016). Performance-oriented transfers, such as EFT, have not yet been implemented in a supra-national governance system.[46]

Summarising the theoretical foundation for a EU-EFT scheme, we argue that

i) positive spill-over benefits from N2k and the realization of EU level interests at decentral levels call for an internalization via fiscal transfers, and

ii) a performance-oriented design would facilitate some decentral spending and implementation autonomy which allows for a greater degree of freedom in realization of decentral level interests.

6.2.2 Current EFT Experiences

The very first EFT scheme was developed in the Brazilian state of Paraná in 1991 where a large share of local tax revenue comes from the generation of value-added taxes (Loureiro, 2002). Before 1991 those municipalities that hosted a large portion of state or federal PA were disadvantaged in terms of foregone income through land-use restrictions imposed by conservation and watershed protection areas. They thus had difficulties to obtain sufficient funds to cover the expenditure of their public functions and required compensation (Grieg-Gran, 2000). An alliance of municipal actors and the state's legislative assembly teamed up for the creation of a fiscal transfer scheme that included ecological indicators alongside socio-economic indicators (Loureiro, 2002). As a result, municipalities that host PAs now receive a share of tax revenue (in the Brazilian case a portion of the value-added tax). While the original idea of the Brazilian EFT was to compensate municipalities for foregone tax revenue, the scheme evolved and transformed into being perceived as an incentive mechanism for conservation (Loureiro, 2002). The novel instrument also spread among other Brazilian states such that currently 17 out of 26 states introduced EFT in their intergovernmental fiscal transfer law (Droste et al., 2017c).

In EFT schemes, as currently in place in Brazil and Portugal, municipalities that host PA receive EFT that have no specific spending purposes attached. While the Brazilian schemes mainly use the share of protected areas in total municipal area in per cent, the Portuguese system mainly uses the total area under protection in the municipality in hectares and only to some extent the share of municipal territory occupied by PA (for a detailed description of the Portuguese EFT system, see Santos et al., 2012, 2015). As these transfers are conditional on the existence of PA, we interpret them as result or performance-oriented transfers. Furthermore, the Brazilian scheme has the advantage to take account of the relative land-use restrictions imposed by PA irrespective of the jurisdictions' size. We thus build our EU-level approach on the Brazilian scheme.

Formalising the Brazilian scheme, an environmental index EI_i is calculated (equation 6.1). Its components are the municipal conservation factor MCF_i (equation 6.2) and the state conservation factor SCF. The MCF_i is given by the sum of m protected

[46] A result-based design of agri-environmental measures (Russi et al., 2016) follows a similar approach but addresses private land users instead.

areas (PA_i) in municipality i's protected areas (PA) and its total municipal area (M). In the calculation of MCF_i differente PA categories are weigthes with w_k according to their contribution to conservation goals, ranging from low weights for less land-use restrictive PA and heavier weights for stricter PA (Grieg-Gran, 2000; Loureiro, 2002; Ring, 2008c). Often the MCF_i is calculated as a percentage. The state conservation factor SCF is defined by the sum of all n municipalities' MCFs. Finally, the EI_i is included as a factor in the allocation mechanism of particular tax revenues and its distribution to local governments.[47]

$$EI_i = \frac{MCF_i}{SCF} \tag{6.1}$$

$$MCF_i = \sum_{j=1}^{m} w_k \frac{PA_j}{M_i} \tag{6.2}$$

$$SCF = \sum_{i=1}^{n} MCF_i \tag{6.3}$$

Based on this original design further EFT reforms have been introduced in Paraná, which led to the inclusion of additional criteria on the quality of the PA (Loureiro, Pinto, and Motta, 2008). This takes into account a second quality criterion beyond the PA category weight w_k, the variation in the quality of the PA or ΔqPA (Loureiro, Pinto, and Motta, 2008). This criterion changes the calculation of MCF_i by adding a weight according to the change in quality of all m PA into the formula for PA of municipality i (equation 6.4). The respective quality changes are assessed yearly (Loureiro, Pinto, and Motta, 2008). In Portugal neither the different categories, nor the quality of PA are taken into account (Santos et al., 2012).

$$MCF_i = \sum_{j=1}^{m} \frac{w_k PA_j}{M_i} (1 + \Delta qPA_j) \tag{6.4}$$

Regarding the effects of EFT, the first econometric policy evaluation studies have been conducted for panel data of the state of Paraná (Sauquet, Marchand, and Féres, 2014), all Brazilian states (Droste et al., 2017c), and Portugal (Droste et al., 2017a). These studies conclude that after introducing an EFT scheme, municipalities respond to the monetary, fiscal incentive inherent in designating a share of tax revenue to ecological indicators such as PA share by the creation of additional municipal protected areas. However, it is important to note that for such a response to an EFT the existence of respective municipal competencies to designate PA on their own is a requirement (Droste et al., 2016). This is a crucial element for the design of similar schemes: only if the addressed jurisdictions have respective competencies in nature conservation policies, the

[47] In Brazil it is a percentage of about up to 5 per cent of the state-level value-added tax. In Portugal, about 5 to 10 per cent of the General Municipal Fund is allocated according to PA location and coverage.

incentive effect may actually result in enhanced conservation efforts (see section 6.2.3 for competencies regarding N2k areas).

In summary, the experiecnes with existint EFT schemes suggest that:

i) they incentivise a positive attitude towards conservation and improve conservation efforts through conditionality on performance for given ecological criteria, while

ii) a respective EU level adaptation would have to take into account the respective conservation competencies of jurisdictions that could receive an EU-EFT scheme, e.g. regarding N2k areas, in order to stimulate a response.

6.2.3 N2k Network Implementation

The European N2k network consists of sites designated under the Habitats Directive and the Birds Directive (Evans, 2012). In total, there are more than $27,000$ sites, covering over 18 per cent of EU terrestrial territory and important marine areas (EU, 2015). In terms of target achievement, mid-term evaluation of the EU biodiversity strategy states that the full implementation of the N2k network exhibits insufficient progress (European Commission, 2015). A recent study regarding the effectiveness and fitness of the Nature Directives found that they are effective "*where they are fully and properly implemented [although] ... there has been limited progress towards improving the status of most European protected species and habitats [and] ... examples suggest that efficiency could be improved by more cost-effective implementation, especially at national and regional level*" (Milieu, IEEP, and ICF, 2016, p. 518). The same study concludes that among the top priority areas for improvement are "*the availability of public funding*" and the management (plans) of N2k sites (Milieu, IEEP, and ICF, 2016, p. 520). In order to synthesise experience with N2k implementation, i.e. regarding the responsibilities and competencies of different government levels, we will therefore review the designation process and the respective competencies of decentral authorities, before discussing financial issues and the suitability of existing EU financing mechanisms for an EFT-like scheme.

Designation Process: Basically, the N2k network has been designated by Member States and/or their respective decentral authorities to protect (migratory) species and ecologically important natural habitats and species (Evans, 2012). After an initial proposal of sites by Member States and an iterative process via conservation seminars on potentially missing habitats and species in which EU officials, observers and environmental NGOs participated, the N2k area list was continuously determined and specified (Evans, 2012). A relatively recent development is the designation of Marine Protected Areas (MPA) within the N2k network (Evans, 2012). While it has been stated that N2k started with a technocratic approach it has broadened over time and includes many more (but not necessarily sufficient) stakeholders by now (Ferranti et al., 2013). This is to say the designation involves expertise from EU conservation officials but also relies upon suggestions and proposals by Member States, their respective decentral levels of government, and civil society organisations. The official designation of N2k sites is, however, within the legal competence of Member States or their sub-national governments.

Implementation Process: Article 6 of the Habitats Directive and Article 4 of the Birds Directive stipulate that appropriate conservation activities have to be realised and deteriorating activities have to be avoided. While the management plans are optional, "necessary" conservation measures have to be implemented by the Member States in conformity with the subsidiarity principle (European Commission, 2014). Thus, generally it is the legal obligation of the Member States to provide designated N2k sites and to ensure the favourable conservation status of species and habitats under protection (for a detailed discussion of the legal meaning of favourable see Epstein et al., 2016). The respective planning and management tasks are, however, often delegated to decentral government levels. Of 24 Member States who replied to a questionnaire about EU conservation measures, 14 explicitly mentioned at least partial decentral management responsibilities for N2k sites (European Commission, 2014, Annex II). In about half of the EU states management plans are obligatory for all N2k sites, in some only for particular sites. Conservation measures include statutory, administrative or contractual measures ranging from specifications of legal activities on-site to contracts between authorities and landowners. Sometimes implementation is performed by NGOs or private landlords (European Commission, 2014, Annex II).

Existing co-financing mechanisms for N2k through EU funds: The establishment and implementation of the Natura 2000 network is mainly financed by the Member States and/or their regional or local authorities although, as provided for in EU nature legislation, co-funding is also available from the EU (Kettunen, Torkler, and Rayment, 2014). A body of evidence shows that there is a substantial gap regarding the finances available for different conservation activities, including the running cost of N2k managing bodies (Kettunen et al., 2011, 2017; Milieu, IEEP, and ICF, 2016; N2k Group, 2016). This financing gap is of crucial importance since a fully operational and effective network of PA requires a range of ongoing management activities such as the restoration of sites. Innovative financing instruments, such as payments for ecosystem services, offset schemes and fiscal incentives like EFT, have been suggested as means to help to bridge the gap (Kettunen, Torkler, and Rayment, 2014; Kettunen et al., 2017). Nevertheless, there is a range of funds and financial sources already available within the EU budget to co-finance the establishment and implementation of N2k in Member States (Kettunen, Torkler, and Rayment, 2014; Kettunen et al., 2017). Both public authorities and private land users can receive EU funding, but their eligibility varies between different funds. In the following, we provide information on the most relevant funds. For a more detailed overview see Table 6.1 in the supplementary material.

The main instrument to fund the promotion of the environment within the EU is the Programme for Environment and Climate Action (LIFE) which aims at a shift towards a resource-efficient, low-carbon and climate-resilient economy, environmental protection and nature conservation. About 40 per cent of the LIFE fund is dedicated to the conservation of nature and biodiversity[48] allocated through applications for project calls (Kettunen, Torkler, and Rayment, 2014). LIFE action grants are available for a wide variety of conservation projects ranging from pilot and demonstration to awareness and dissemination projects. However, most of the other EU funds contribute to N2k policies at least to some extent due to the cross-cutting, integrative nature of EU nature and environment policies (cf. Kettunen et al., 2017, see also table Table 6.1).

The suitability of existing EU funds for the integration of an EFT-like mechanism: Regarding the choice of a suitable EU fund that could implement a potential EU-EFT mechanism, most of the funds can be dismissed due to their narrowly defined purposes that do not allow for a broad and large-scale integration of performance-based conservation transfers addressing public authorities (see the Table 6.1 for an overview of EU funds an their purposes). The European Maritime and Fisheries Fund (EMFF) finances fishery and coastal policies and it thus not suited for terrestrial N2k. The European Agricultural Rural Development Fund (EARDF) and thus the Common Agricultural Policy (CAP) targets agriculture and forestry related activities. While EARDF is a key fund to finance concrete conservation actions in these sectors, biodiversity conservation and N2k are not restricted to those types of land use and have to be mainstreamed further. Thus, only the LIFE fund and the European Fund for Regional Development (ERDF) are remain as potential options. Both have conservation activities[49] such as N2k implementation as their explicit goals, finance related activities, and incentivise behavioural changes within public administrations.

One option would be to increase LIFE such that sufficient funding is available at the implementing government levels. From a public finance perspective, such specific-purpose transfers (with matching or co-financing conditions) are seen as the most appropriate in order to ensure sufficient funding and internalise spill-over benefits of the realisation of higher government level interests. From the perspective of effectiveness, evidence confirms that LIFE funding is generally effective in delivering conservation outcomes (Kettunen et al., 2017; Milieu, IEEP, and ICF, 2016). Thus, for helping to close the N2k financing gap LIFE can be considered the appropriate EU fund via an augmentation of available co-finance. Nevertheless, for performance-oriented funds without spending specifications the ERDF is more suitable. The ERDF finances Operational Programmes (OP) defined by Member States or their sub-national jurisdictions. Portions of the programmes are spent on pre-specified priority areas (depending on economic development stages), which provides substantial regional spending autonomy. Since EFT normally allocate public revenue according to specific ecological criteria such as PA coverage but without earmarking for particular spending purposes,

[48] Around 75 per cent of total LIFE funding is allocated to the sub-programme for Environment, of which at least 55 per cent of the resources dedicated to projects financed by way of action grants shall be allocated to support the conservation of nature and biodiversity.

[49] The ERDF regulation explicitly mentions an indicator on "surface area of habitats supported in order to attain a better conservation status" (EU Regulation 1301/2013, Annex 1).

the ERDF seems the most appropriate EU fund whose support to biodiversity conservation could be enhanced through an EFT-like scheme. This is particularly the case since, despite of efforts, ERDF's contribution to supporting biodiversity objectives remains limited (Kettunen et al., 2017; Milieu, IEEP, and ICF, 2016; N2k Group, 2016). However, a prime goal of ERDF is economic and social convergence among EU regions which is also called cohesion policy. Thus, preference is given to less developed Member States and remote, mountainous or sparsely populated regions. For an integration of an EU-EFT scheme within the ERDF, resulting allocative patterns would have to be in line with the cohesion policy and mostly benefit economically marginalised regions.

Summarising the experience and evaluations of N2k progress, this means that:

i) an improvement of the implementation of the Habitats and Birds Directives in terms of ecological effectiveness and cost-effectiveness is needed both at national and regional/local level;

ii) (environmental) ministries and decentral government levels, or the two of them conjointly, are among the key responsible authorities for the implementation of the network and any improvement mechanisms would have to address them;

iii) financing opportunities for biodiversity also exist at EU level, but both the current national and EU schemes are not sufficient to meet N2k financing needs;

iv) a performance-oriented EU-EFT mechanism could be integrated into the ERDF in order to complement the existing but insufficient funding structure for N2k through setting additional incentives while maintaining implementation autonomy.

6.3 Ecological Fiscal Transfers at EU level – Proposing Performance-Oriented Transfers

The design for a European EFT scheme proposed and simulated in the context of this article (see below) is an adaptation of the original scheme in Paraná, Brazil. The Paraná scheme is the most mature EFT mechanism to date, including a continuous improvement of the scheme over time (Loureiro, 2002; Loureiro, Pinto, and Motta, 2008). Furthermore, in addition to PA coverage the scheme also takes into consideration variations in PA quality (see section 6.2.1).

In order to design an EU-EFT scheme that creates an incentive for conservation efforts and improves conservation outcomes, we propose a scheme composed of two main parts: one quantitative and one qualitative measurement. While the first and quantitative part measures the relative area under N2k protection and thus incorporates the corresponding fiscal needs, the second measures conservation management outcomes in terms of the portion of habitats with a favourable conservation status and is thus based on conservation performance. Regarding potential incentive effects under such an EU-EFT scheme, those jurisdictions that can increase N2k coverage through additional designations would be incentivised to do so, while those that only have PA management competencies would be incentivised to improve their N2k site management quality efforts (see section 6.5 for a more detailed discussion of potential effects of an EU-EFT mechanism). Formally, the allocative rule can be expressed as jurisdiction's i portion of a $fund$ distributed among all j to n jurisdictions (equation 6.5)

$$\mathrm{EFT}_i = \left(\frac{\mathrm{CF}_i}{\sum_{j=1}^{n} \mathrm{CF}_j}\right)\mathrm{fund} \tag{6.5}$$

where

$$\mathrm{CF}_i = \frac{\mathrm{PA}_i}{\mathrm{area}_i} + \frac{\mathrm{FCS}_i}{\mathrm{habitats}_i} \tag{6.6}$$

the Conservation Factor (CF) is determined by the sum of the share of Protected Area expanse (PA) in total area (area) in per cent, and the number of habitats with favourable conservation status (FCS) as a per cent share of the total number of reported habitats (habitats) for each jurisdiction i (equation 6.6).[50] For the subsequent analysis we simulated an EU-EFT mechanism according to the above-described allocative criteria (see section 6.4).

Concerning the feasibility of such a mechanism, given available data there is a constant monitoring of N2k sites (EU, 2015), and Art. 17 of the Habitats Directive and Art. 12 of the Birds Directive require regular quality assessments of the respective N2k habitat statuses and species developments. While quality monitoring is currently due every six years and has been reported twice, the reporting frequency could theoretically be increased once sufficient and standardised institutional knowledge has been acquired.

6.4 Empirical Patterns – Who Would Benefit?

In order to assess who (i.e. which areas) would be the beneficiaries of a potential EU-EFT scheme, we analyse the empirical patterns of the spatial distribution of N2k areas among EU-27 NUTS 2 regions. NUTS is the nomenclature of territorial units for statistics (*Nomenclature des unités territoriales statistiques*). The nomenclature subdivides Member States hierarchically and references the units by a geocode.[51] The system is

[50]The favourable conservation status refers to habitats found within N2k sites but to all habitats within a regions territory.

[51] The code starts with a two letter code referencing the uppermost level of Member States. Each of the following levels is identified by a single numeral (plus a letter in case there are more than 9). NUTS 1 are major economic regions such as regions, states, provinces or groups of them. NUTS 2 are basic regions for the application of regional policies such as counties or planning, territorial or government regions – depending on the Member State. NUTS 3 are small regions for specific diagnoses and may be represented by districts, prefectures or counties. Regarding policy, regions "eligible for support from cohesion policy have been defined at NUTS 2 level" (Eurostat, 2016), which therefore provides a suitable data basis for simulating a potential EU-EFT mechanism.

used in the EU for statistical and analytical purposes but also plays a crucial role in framing EU policies and allocating EU funds, i.e. for ERDF, ESF and CF (Eurostat, 2016). For each Member State there are 3 NUTS levels – which means the NUTS structure is closely related to the administrative structure of the Member States but is not necessarily identical. By structuring our analysis according to NUTS 2 regions we have readily available statistical data and can estimate socio-economic characteristics of regions that host PA. Furthermore, we can conduct assessments at a regional level that is closely related to the distributive mechanism of EU funds, such as the ERDF (see section 6.2.3 for more detail).

6.4.1 Data Sources, Preparation and Software

The N2k data was retrieved from the European Environment Agency (2015) as shapefiles for the years 2009-2013. From these files the intersection with 2013 NUTS 2 regions (Eurostat, 2015) has been tabulated with a proprietary GIS software such that the percentage of N2k area per NUTS 2 region and year was calculated. In order to assess socio-economic characteristics of the potential beneficiaries, data for $2009 - 2013$ on the NUTS 2 regions' area (in km^2), GDP per capita (regional gross domestic product in purchasing power standard per inhabitant), population density (persons per km^2), tourism (nights spent at tourist accommodation establishments), and unemployment (unemployment rate in per cent) was retrieved from Eurostat (2015). Moreover, to assess bio-physical characteristics of potential beneficiaries, the percentages of the NUTS 2 regions in bio-geographical regions as delineated by the Habitats Directive were computed[52] based on European Environment Agency data (2015). Summary statistics can be found in the appendix and plotted maps of these variables in the supplementary material (figures 6.3 – 6.6). A dataset for the EU-27 NUTS 2 regions was constructed and overseas regions were excluded.[53] Since there were fractions of missing data and this would have led to a large overall loss of information within regressions, missing observations were imputed[54] with the ***Amelia*** package (Honaker, King, and Blackwell, 2011) in the **R** environment (R Development Core Team, 2016)specifying lower and upper limits (0.001 and 1.2 times the maximum observed values) and imposing a linear time trend. Missing values were imputed 100 times and these data sets were used to average the imputations. This resulted in a single balanced panel data set with $n = 266$ EU-27 NUTS 2 regions, and $T = 5$ years of observation. The observations for the proportion of habitats in favourable conservation status (European Environment Agency,

[52] To eliminate inaccuracies in cropping the polygons we re-classified greater or equal 99 per cent shares as 100, and less or equal to 1 per cent as 0. The map of bio-geographical regions can be found in the supplementary material.

[53] Excluded were EU-27 NUTS 2 regions that are geographically located on other continents due to their extra-continental statuses: ES70, FRA1, FRA3, FRA3, FRA4, FRA5, PT20 and PT30. Furthermore, Croatia (HR03, HR04) has not been integrated since it became an EU Member State in 2013.

[54] The fractions of missing values that were imputed are: N2k (0.032), area (0.012), population density (0.011), GDP per capita (0.098), tourism (0.108), unemployment (0.013), and proportion of favourable conservation status (0.005).

2015)were only available for the year 2013 (for the reporting period of $2008 - 2012$). The dataset was thus reduced to a 2013 cross-section subset with $n = 266$ EU-27 NUTS 2 regions.[55] The maps (see section 6.4.3) have been produced with a combination of the **R** packages ***sp*** (Pebesma and Bivand, 2005), ***maptools*** (Bivand and Piras, 2015) and ***rworldmap*** (South, 2011). Additionally, some functions from ***spdep*** (Bivand and Piras, 2015) have been employed. For the analysis of the spatial distribution of EU-EFT flows on the 2013 subset, a regresssion tree model was used which was supplied by the ***rpart*** package (Therneau and Atkinson, 2015) and trained through cross-validation with the ***caret*** package (Kuhn, 2008; Kuhn et al., 2016). In addition, a random forest model was estimated for robustness checks of the decision tree (Liaw and Wiener, 2002). The summary table has been produced with stargazer (Hlavac, 2015).

6.4.2 Econometric Model

In order to analyse where the EFT would flow and in order to account for interactions and non-linearities, we employ a classification or "*decision tree*" model (Hastie and Tibshirani, 2009, chap. 9.2). Tree-based methods partition (multidimensional) data into clusters, groups or regions. The greedy algorithm, also known as recursive binary splitting, proceeds as follows to grow a regression tree (Therneau and Atkinson, 2015). At the first internal node the entire data is split into two regions such that the Residual Sum of Squares is minimised, which means the variable and cut point with the greatest predictive power is chosen. Resulting groups are characterised by statistically significant different averages of the dependent variable; say the left-hand branch has a low average and the right-hand side a high average. The splitting process is repeated for each of the resulting branches until no further gain in explanatory power can be obtained through additional splits. The terminal nodes or leaves of the tree represent the resulting regions or partitions with different average response variable values. For our analysis we use a regression tree with the following structure (equation 6.7).

$$\begin{aligned} \text{EFT}_i &= \text{area}_i + \text{popdens}_i + \text{GPDcap}_i \\ &+ \text{tour}_i + \text{unemp}_i + \text{bioregions}_i + \epsilon_i \end{aligned} \tag{6.7}$$

where EFT is monetary flow of EFT payments for region that are allocated among EU-27 NUTS 2 regions based on the proposed design (see section 6.3) of an arbitrarily chosen fund size of € 1 billion, area is the area in km^2, popdens is persons per km^2, GDPcap is the GDP per capita in PPS (purchasing power standard), tour is the overnight stays in tourist accommodation establishments, unemp is the unemployment in per cent, bioregions is vector of variables measuring the share of area in the

[55] Covariate data for 2013 is not complete for all NUTS 2 regions. For the missing ones, there is data for previous years – which facilitates an imputation of missing data for 2013 such that we have one complete set of observations for the year for which conservation statuses are reported.

respective bio-geographical regions Alpine (ALP), Atlantic (ATL), Black Sea (BLK), Boreal (BOR), Continental (CON), Mediterranean (MED), Pannonian (PAN) and Steppic (STE), and ϵ is the residual error term. To avoid overfitting, we pruned the tree with a complexity parameter obtained by a tenfold cross validation (Kuhn et al., 2016). Each variable is observed (or imputed, see section 6.4.1) for EU-27 NUTS 2 regions for 2013. In order to check for robustness we also employed a Random Forest model that repeatedly grows decision trees and thus allows to average over the ensemble of multiple trees (Liaw and Wiener, 2002). The corresponding variable importance plot can be found in the supplementary material. At this point it suffices to say that the variables included in the presented decision tree are among the most important ones given a tenfold cross-validated ensemble of $10,000$ trees.

6.4.3 Spatial Distribution of N2k Sites in EU-27

Figure 6.1 displays the different components of the proposed EU-EFT design: namely the quantitative part (percentage of N2k coverage in NUTS 2 regions), the qualitative part (proportion of reported habitats that are in favourable conservation status), and the distributional pattern of finances that would be allocated through the proposed mechanism. While N2k coverage is stronger in Southern and Eastern Europe, there are a couple of regions with a greater proportion of reported habitats that were assessed with a favourable conservation status: northern Sweden, Slovakia, Romania, Southern Germany, Austria, Slovenia, Southern France, Italy, and Southern Portugal. The payments that result from the proposed EFT design are relatively even in their distribution with low payments in the Atlantic region, Poland and Czech Republic, and top payments in Cyprus, Romania, Slovenia, Slovakia and Malta. A histogram of the payments can be found in the supplementary material.

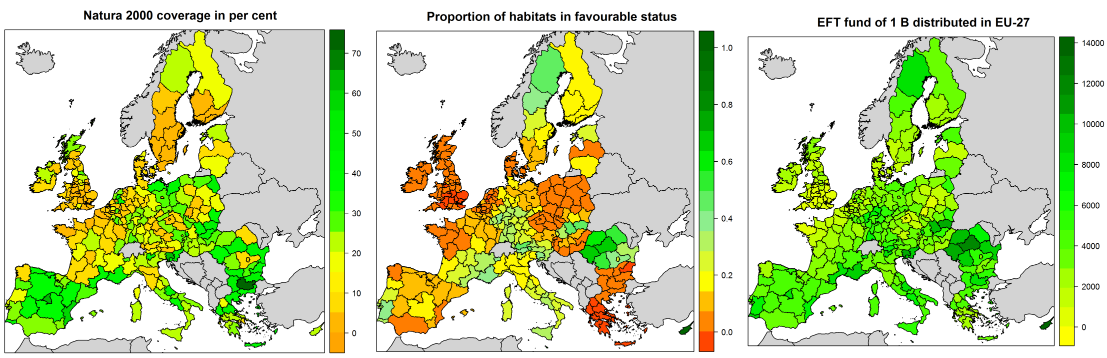

FIGURE 6.1: Spatial distribution maps: a) Percentage of EU-27 NUTS 2 regions' area covered by N2k sites in 2013; b) proportion of habitats in favourable conservation status as reported by EU-27 Member States under Article 17 of the Habitats Directive for the 2008 − 2012 period; c) distributional pattern of the proposed EU-EFT mechanism for an arbitrary quantity of €1 billion. *Source:* authors' computation based on European Environment Agency (2015).

Figure 6.2 displays a regression tree, where the EFT payments have been clustered. At each node it splits the data further into subgroups and the final nodes or leaves display the average payment in that particular group. The left branches correspond with a true condition. The tree starts with a split on the Atlantic bio-geographical region, at greater or equal to 40 per cent of the NUTS 2 regions within that region. Together with the second node for the Atlantic regions at greater or equal to 99 this reads: if a NUTS 2 region is 100 per cent in the Atlantic region, it would on average receive € $1,434,000$ out of a € 1 billion EU-EFT fund. If it has between 40 and 99 per cent of its area in the Atlantic region, it will receive on average € $2,846,000$. The third node splits at less than 3 per cent in the Alpine region and continues with splits for the Meditarranean region, tourist overnight stays, unemployment rates and GDP per capita. For the non-Alpine, Mediterranean regions, the regions with high unemployment on average receive less EFT payments than the ones with lower unemployment. The touristically attractive Alpine regions receive on average high payments but less than the less touristically developed ones. The highest payment is received by Alpine regions that have a GDP per capita less than € $18,000$. On average this is to say that remote mountainous and economically poor regions would receive the highest EFT payments – which would qualify the proposed EFT mechanism to be in line with the cohesion policy of the ERDF.

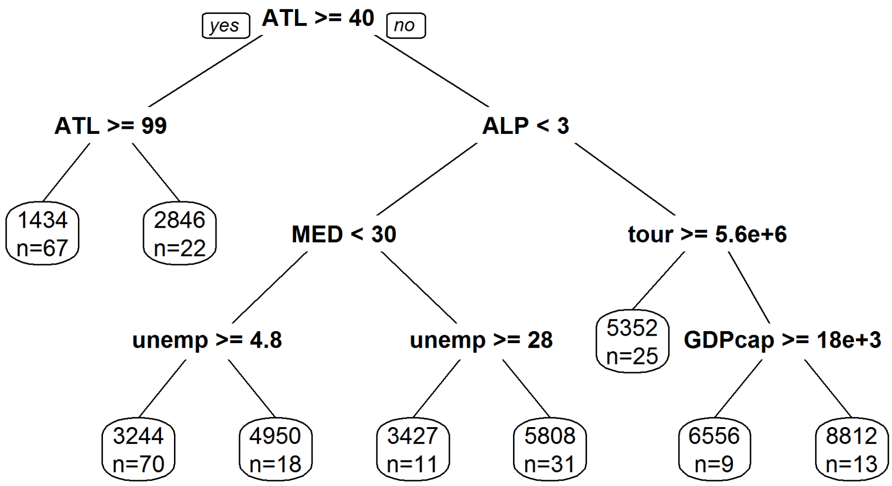

FIGURE 6.2: A regression tree for the proposed EU-EFT mechanism, showing to which regions the EFT would flow (final sums are based on an arbitrary 1 billion EFT sum, numbers are given in € 1,000), each node's decision variable and its partitioning is given in bold, and the variables are: Atlantic (ATL), Alpine (ALP) and Mediterranean (MED) bio-geographical regions, overnight stays in tourist accommodation establishments (tour), unemployment in per cent (unemp), and GDP per capita (GDPcap). Source: authors' computation based on European Environment Agency (2015) and Eurostat (2015).

6.5 Discussion – Criteria for Evaluating Outcomes of the Proposed Scheme

When evaluating the effect of integration of ecological indicators into the allocative rules of EU funds, an environmental policy analysis from a public finance perspective may consider three key aspects: ecological effectiveness, the distribution of income, and the efficiency in resource allocation. As a basic condition the following arguments to hold, the EU-EFT fund size would have to be sufficiently large. If the resulting payments are higher than (opportunity) costs for either N2k designation or improved management quality, the resulting incentives would most likely have a considerable effect on biodiversity conservation efforts. However, success in terms of improved biodiversity conservation is a sum of all EU and national funds combined. Even a slightly improved ERDF situation could help to lift that overall success level since is reduces costs for N2k and thus raise the stakeholder willingness from the current level.

6.5.1 Ecological Effectiveness

The ecological effectiveness of the EU-EFT could be measured as its contribution to attaining EU biodiversity and conservation goals. In such terms the result of our proposed EFT mechanism would strongly depend on both i) the robustness of the ecological indicators and ii) the governance structures in place. We propose that the EU-EFT scheme is composed of a quantitative indicator measured by terrestrial N2k coverage of the NUTS 2 regions and a qualitative indicator measuring the proportion of habitats reported according to the EU Nature Directives that were assessed with a favourable conservation status. Assuming that the resulting financial flows would actually set an incentive for the regions to enhance N2k sites and/or conservation management such that both or one of these indicators rise, one could expect a significant contribution to the EU conservation goals. Such an assertion, however, depends on two factors that limit the certainty of predicting outcomes: the importance of the chosen indicators for attaining the EU conservation goals and the ability of the regions to provide the required actions.

In terms of the first, the N2k sites themselves contribute to attaining multiple goals set in the EU biodiversity strategy. In target 1, action 1, it is mentioned that the N2k network is to be completed and that further species and habitats are to be integrated within and beyond N2k networks (European Commission, 2011). The mid-term review states that the "*Natura 2000 network has been largely completed for* ***terrestrial and inland water habitats***, *covering about 18% of the land surface*" (European Commission, 2015, bold by authors).[56] Thus there seems to be some but no great political demand

[56] The mid-term review of the EU biodiversity strategy also states that "The marine network coverage has increased to 6%, still well below the 10% global target" (European Commission, 2015), which might require an inclusion of marine N2k sites into the proposed EFT mechanism which has so far not been possible due to data limitations.

for more N2k sites. However, the mid-term review also states that the goal of securing and improving a defined percentage of species', birds' and habitats' conservation status shows an insufficient rate of progress and that increased efforts are required.[57] Assuming that N2k sites have to be appropriately established and managed to help secure a good conservation status of threatened species and habitats (cf. Gruber et al., 2012, for species protection gaps), the proposed EFT mechanism might help to attain such a conservation goal, especially since our EU-EFT proposal consists of a qualitative part which addresses the proportion of favourable conservation status directly. Considering that N2k management is more likely to be under the authority of decentral governments and that the ratio of favourable conservation status habitats is low, the qualitative indicator could be given a stronger weight than in our current policy design. In this context, it is crucial to ensure appropriate monitoring of species and habitats so that a reported improvement in conservation status is not just an improvement on paper. Further work on the issue should also include marine PA.

Secondly, a fiscal incentive will only lead to an effect if the targeted jurisdictions have competencies that can correspond to the incentive (see section 6.2.2). While we have reviewed Member States' responses about management responsibilities (European Commission, 2014, Annex II) and found that a majority of respondents has indicated at least a partial responsibility of decentral public authorities at NUTS 2 level (see section 6.2.3), this is not necessarily representative. In most Member States, national authorities such as ministries or at least NUTS 1 regions are responsible for the planning and designation of N2k sites. Thus our assumption that – through the implementation of an EU-EFT mechanism – incentives are creates for both the designation and the management quality of N2k sites depend on the different government and governance structure of the Member States. Since there is an institutional learning and N2k decisions are increasingly organised in a participatory fashion (Ferranti et al., 2013), we would expect that on average decentral authorities have at least some right to say in the respective planning and/or management procedures. For a more certain response, a comparative study of the exact decision-making competencies and planning procedures for N2k sites and management remains a future research question.

6.5.2 Distributive Effects

In the case of the EU-EFT model that we proposed (section 6.3) the transfers would be allocated according to the share of N2k area in the jurisdictions' territory and the proportion of reported habitats in favourable conservation status. According to our

[57] The potential outcome of an EU-EFT in terms of total protected area might be stronger if design was not to assess the N2k share per region area but rather total N2k area per region. The incentive would thus be stronger for regions with a larger territory. At the same time an equally distributed habitat network would rather require that PA can be found among all regions not mainly large ones. Depending on the conservation goal, both designs would have their merits. Future work may compare different design options.

assessment of the spatial distribution of resulting EFT payments (section 6.4), the highest sums would flow to mountainous and economically weak regions – which are very likely disadvantaged ones. The lowest payments would on average flow to the Atlantic region – which is an economically strong region in Europe that neither supplies great proportions of its area as N2k sites nor has a high ratio of favourable statuses and reported habitats (see figure 6.1). If the ERDF, is allocated differently through an EFT mechanism, there would likely be losers in comparison to the status quo who are not eligible for receipt of that part of the funds any longer. The beneficiaries, mainly remote mountainous and economically less developed NUTS 2 regions (see section 6.4.3), however, would be well aligned with the cohesion policy goal of the ERDF. The resulting payments could be used by the decentral governments on any field they see necessary. An EU-EFT scheme within the ERDF would allocate a share of it according to the ecological indicators employed. This turns N2k sites and habitat quality into a source of income. If activities are pursued that diminish the quality of habitats that would reduce transfers of an EU-EFT. These incentives would balance the ERDF's main dedication to economic development without counteracting regional spending autonomy and thus help mainstream biodiversity.

But there remains an important element in the distributional effects of our EU-EFT proposal with respect to the chosen ecological indicators. Our proposal contains a quantitative part, which is fiscal-need-based in the sense that those regions with the highest N2k share and related costs would receive higher transfers. The qualitative part which measures performance in terms of habitat qualities is performance-based and thus sets a behavioural incentive to provide a better quality (see Burton and Schwarz, 2013; Muradian, 2013; Russi et al., 2016, for comparable literature on incentives in agri-environment measures and ecosystem services based payments). Regarding conservation efforts, both early action (a high N2k share) and outcomes of recent action (habitat quality) will thus be rewarded under the suggested EFT scheme.

In a nutshell, the proposed EFT mechanism benefits those who provide desired results in terms of EU conservation policies such as N2k site coverage and favourable conservation status of habitats – which according to our analysis are mainly rather remote, economically and touristically less developed. mountainous regions. Therefore, the proposed EU-EFT scheme within the ERDF mechanism would be in accordance with EU cohesion policy – which is a prerequisite for an EU-EFT to be integrated in the ERDF (see section 6.2.3).

6.5.3 Cost-Effectiveness

In terms of cost-effectiveness or least-cost provision, it matters which EU conservation goals are set, e.g. within the EU biodiversity strategy, and at what cost they can be attained. In this context there is one particularly important differentiation between refinancing fiscal needs for conservation and stimulating performance.

As regards closing an N2k financing gap, it is therefore important to consider that due to the performance-oriented transfer design without specific spending conditions EFT mechanisms are not an instrument to refinance conservation needs *directly*. The revenues received can, but do not have to, be spent on conservation. For closing a specific financing gap dedicated specific-purpose funds are considered more suitable (Kettunen et al., 2011, 2017; Milieu, IEEP, and ICF, 2016; N2k Group, 2016). Increasing specific-purpose funds, such as LIFE, would be better suited to help to close the financing gap in a targeted manner. Our EU-EFT is partly fiscal-needs-based in so far as it includes a quantitative N2k area proportion which incurs costs. At the same time it also has no spending purposes attached and may thus not fully serve to ensure an effective implementation of the Nature Directives. For these reasons, an EU-EFT scheme may only serve as a complement but not a substitute for existing direct and earmarked biodiversity funding mechanisms such as LIFE.

However, our performance-oriented approach has the benefit of cost-effectively incentivising a greater willingness to increase conservation efforts on the part of the addressees. Considering the functioning of EFT from a perspective of rational decision making (which might not resemble the complete picture), especially those jurisdictions likely react to the incentive that have opportunity and/or (EU co-financed) implementation costs lower or equal to the (non-)financial benefits of enhancing their N2k area or quality. Given a policy goal, say target 1 of the EU biodiversity strategy to fully implement the Nature Directives, the goal could be attained at lowest total costs, since its implementation is realised where it is cheapest. In this sense, an EFT-EU scheme can be considered a cost-effective approach to attain the politically set conservation targets, similar to a standard-price approach (Baumol and Oates, 1971). Beyond this incentive effect our approach induces a benchmarking or yardstick competition regarding conservation performance in both quantitative and qualitative terms: as there is only a limited EU-EFT fund available, the mechanism introduces a dynamic competitive environment among regions for increasing N2k performance over time. Assuming a constant fund size, a region would lose compared to a previous period if other regions increased their performance because transfers for those regions would increase and thus payments for a non-improving regions decrease. Beyond the incentive created through instituting a fiscal transfer for hosting N2k this mechanism would continuously call for performance improvements. Therefore, an EU-EFT may lead to an increase in N2k coverage over time without the need greater expenditure.

6.6 Conclusion – Considering the Political Economy of Conservation

n order to support biodiversity conservation efforts in the EU, we have proposed a design option of a European EFT mechanism to set further incentives for nature conservation efforts of EU regions. We have synthesised current experience with both EFT schemes and the EU-level N2k financing mechanisms and, building on that knowledge and evidence, developed a possible EFT design for the EU level. We have concluded that an EFT-EU scheme would be the best suited to be implemented as a part of the ERDF and it could enhance the current ERDF allocation mechanism through performance-oriented payments based on both quantitative and qualitative N2k indicators without specified conditions on spending the transfers received. To assess potential effects we have simulated the resulting financial flows and analysed the spatial distribution among socio-economic and bio-geographical characteristics of receiving regions. Thereby we provide the first design proposal for an EFT scheme for adaptation beyond the national context. We have provided quantitative simulations to support the analysis and assess potential outcomes of the developed scheme.

The main innovative feature of including of an EU-EFT scheme into the existing mix of EU biodiversity financing instruments is the performance orientation of the transfers without spending conditions. Transfers would flow to regions that supply most (or best managed) N2k sites. As such the scheme would represent a valuable complement to current EU mechanisms for funding biodiversity by balancing funding primarily focused on socio-economic development with incentives for the realisation of EU-wide biodiversity conservation performance and thus help mainstreaming biodiversity. The lack of spending conditions allows for a certain degree of autonomy of the receiving public authorities but the conditions on receiving the EFT create an additional incentive for a cost-effective increase in the quantity and/or habitat quality of N2k sites. Furthermore, EU-EFT would mainly benefit remote and poor mountainous regions and would thus be in line with the cohesion goal of the ERDF. It could help to enhance the currently limited contribution of ERDF to conservation objectives, reinforcing the uptake of existing opportunities to fund biodiversity in the context of national and regional Operational Programmes.

In this respect, the effective functioning of the proposed mechanisms' incentive depends on the actual competencies and decision-making power of regional authorities regarding N2k implementation. Thus, the political economy of conservation in Europe also matters substantially for the outcome of an EU-EFT scheme. In multi-level biodiversity governance systems earmarking is politically not always easily acceptable. An EU-EFT scheme building on general-purpose transfers – while not the most effective in terms of bridging the N2k funding gap – has the potential benefits of being both more politically acceptable and balancing the current primary focus of the ERDF with ecological criteria that create an incentive for biodiversity conservation. For example, if both quantitative and qualitative N2k indicators are assessed based on regional performance, this introduces a yardstick competition for a EFT fund from which mainly remote mountainous and economically less developed regions would benefit. The yardstick competition furthermore sets incentives for continuous performance improvements. Thus the EU-EFT scheme can be seen as a step in the right direction, setting fiscal incentives for conservation, and playing the long game by aiming to subtly change attitudes towards conservation through biodiversity mainstreaming. Within the EU context, future research directions may include a comparative study of the exact decision-making competencies and planning procedures for N2k sites and management in order to specify a corresponding implementation of an EU-EFT mechanism further. Beyond the EU context an adaptation to other multilevel contexts such as federalist states or international/supranational bodies may pose interesting research questions.

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Appendix. Descriptive Statistics

TABLE 6.1: Summary statistics

Statistic	N	Mean	SD	Min	Max
Ecological fiscal transfer (*EFT*) in €1,000	266	3,759.4	2,508.4	2.6	13,364.3
area in square km (*area*)	266	16,280.1	21,936.1	13.4	226,785.4
population density (*pop*)	266	480.5	1,251.3	3.4	10,438.2
GDP per capita (*GDPcap*)	266	25,802.9	10,443.8	8,000.0	86,400.0
tourist overnight stays (*tour*)	266	9,764,382	11,438,209	26,378	77,692,454
unemployment rate (*unemp*)	266	10.5	6.7	2.5	36.2
Alpine region (*ALP*)	266	7.0	19.8	0	100
Atlantic region (*ATL*)	266	32.5	45.2	0	100
Black Sea region (*BLK*)	266	0.2	2.0	0	22
Boreal region (*BOR*)	266	5.3	21.8	0	100
Continental region (*CON*)	266	33.3	42.9	0	100
Meditarranean region (*MED*)	266	17.6	36.9	0	100
Pannonian region (*PAN*)	266	3.5	17.0	0	100
Steppic region (*STE*)	266	0.4	4.7	0	71

Source: authors' computation based on European Environment Agency (2015) and Eurostat (2015). Monetary values are in purchasing power standards (PPS) per inhabitant except for EFT payments which are based on an arbitrary fund size and rather stand for distributive patterns.

Supplementary Material

In this supplementary material we provide details on:

i) existing EU funds in relation to N2k co-financing (Table 6.2)

ii) spatial distribution of socio-economic control variables (Figure 6.3)

iii) biogeographical regions in EU-27 (Figure 6.4)

iv) the frequency of resulting EU-EFT transfers (Figure 6.5)

v) variable importance in the supportive random forest model (Figure 6.6)

Further information regarding the data compilation and statistical analysis can be found at https://github.com/NilsDroste/EFT-EU.

TABLE 6.2: Overview of existing EU funds in relation to N2k co-financing

abbreviation	full name	major objective	eligible areas	key adressees	type of N2k related activities	funding mechanism
LIFE	European financial instrument for the environment	protecting and improving the quality of the environment and halting and reversing biodiversity loss, including support of the Natura 2000 network and tackling the degradation of ecosystems	all Member States	public authorities, private land users and practice-oriented research & development	environment and resource efficiency, nature and biodiversity and environmental governance; climate change mitigation, climate change adaptation and climate governance, information	about 40 per cent of LIFE resources are dedicated to project action grants in support of nature and biodiversity conservation
EARDF	European Agricultural Fund for Rural Development	competitive agriculture; sustainable natural resource management; balanced territorial development	all Member States	farmers / private land owners and -users, public authorities and research & development	restoring, preserving and enhancing ecosystems related to agriculture and forestry	conditional on rural development programmes that address four of six EARDF priorities with at least 30 per cent expenditure related to the environment (climate, forest, organic farming and N2k)
ERDF	European Fund for Regional Development	investment in SMEs; sustainable jobs; investment in energy, environment, transport, ICT, social, health, research, innovation, business, and educational infra- structure; networking, cooperation and exchange of experience	preference for less developed Member States and marginalised regions (remote, mountainous or sparsely populated areas)	public authorities, research & development and private land users	promoting climate change adaptation, risk prevention and management, including supporting investment for adaptation to climate change, including ecosystem-based approaches; preserving and protecting the environment and promoting resource efficiency including protecting and restoring biodiversity and soil and promoting ecosystem services, including through Natura 2000, and green infrastructure	ERDF project implementation in the Member States and regions through operational programmes; support available depends on the level of economic development in terms of per capita GDP

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TABLE 6.2: Overview of existing EU funds

abbreviation	full name	major objective	eligible areas	key adressees	type of N2k related activities	funding mechanism
EMFF	European Maritime and Fisheries Fund	Competitive & sustainable fishery; implementation of fisheries policies; balanced territorial development	all Member States	fishermen, public authorities and research & development	protection and restoration of aquatic biodiversity, enhancement of ecosystems related to aquaculture, and promotion of resource-efficient aquaculture	fund allocation to Member States according to fishing industry size, Member State development of operational programme to be approved by EC with joint implementation
ESF	European Social Fund (ESF)	sustainable and quality employment; labour mobility; social inclusion; poverty reduction; education, training and vocational training; institutional capacity of public authorities	preference for less developed Member States and marginalised regions (remote, mountainous or sparsely populated areas)	societally disadvantaged, such as unemployed or disabled people, over all a wide variety of target groups	enhancement of institutional capacity of public authorities and stakeholders and investment in institutional capacity and in the efficiency of public administrations and public services	ESF project implementation in the Member States and regions through operational programmes; support available depends on the level of economic development; co-financing rates vary between 50 and 85 per cent depending on per capita GDP
CF	Cohesion Fund	Investment in the environment, sustainable development and energy	Member States with a gross national income (GNI) measured in purchasing power parities less than 90 per cent of the average GNI EU-27	public authorities, research & development and private land users	promoting climate change adaptation, risk prevention and management, including supporting investment for adaptation to climate change, including ecosystem-based approaches; preserving and protecting the environment and promoting resource efficiency including protecting and restoring biodiversity and soil and promoting ecosystem services, including through Natura 2000, and green infrastructure	CF project implementation in the Member States and regions through operational programmes; support available depends on the level of economic development in terms of per capita GDP

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TABLE 6.2: Overview of existing EU funds

abbreviation	full name	major objective	eligible areas	key adressees	type of N2k related activities	funding mechanism
FP7 & H2020	Framework Programmes for research and innovation	transnational research in a range of priority areas	all Member States and partly associated states	mainly science and SMEs	Research-related conservation activities	calls for pre-specified research projects

Source: authors' compilation based on Kettunen et al. (2014; 2017) and respective EU legislation.

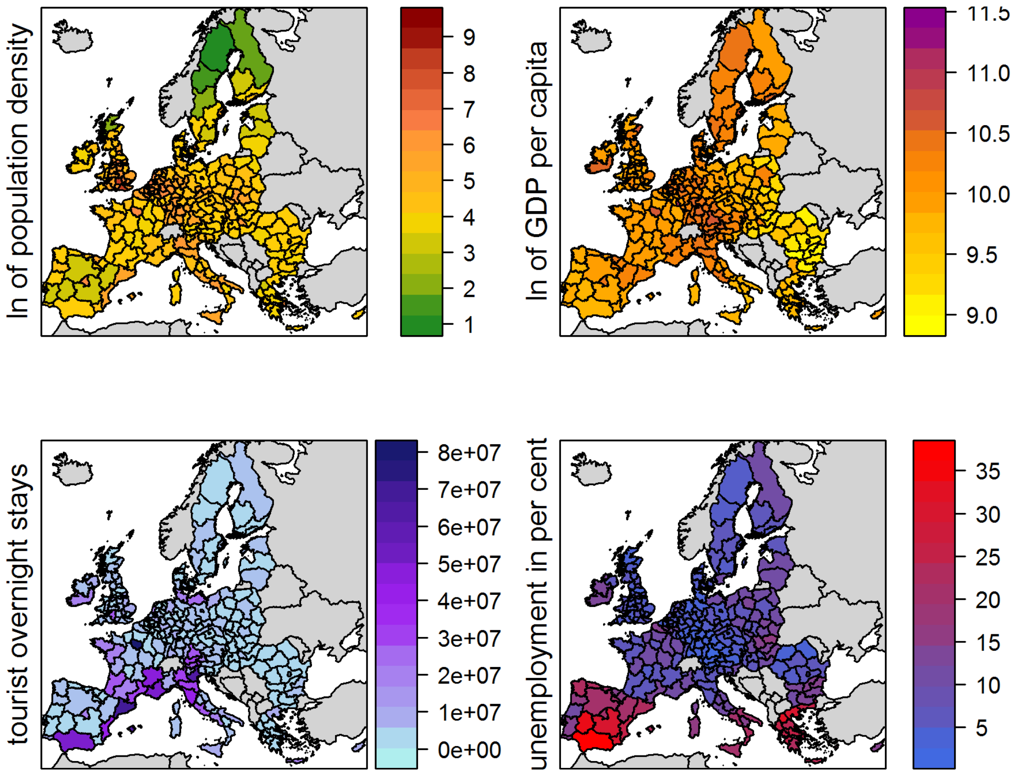

FIGURE 6.3: Spatial distribution of socio-economic control variables for 2013 (log of population density, log of GDP per capita, tourist stays and unemployment rates; from top left to bottom right). *Source:* authors' computation based on Eurostat (2016).

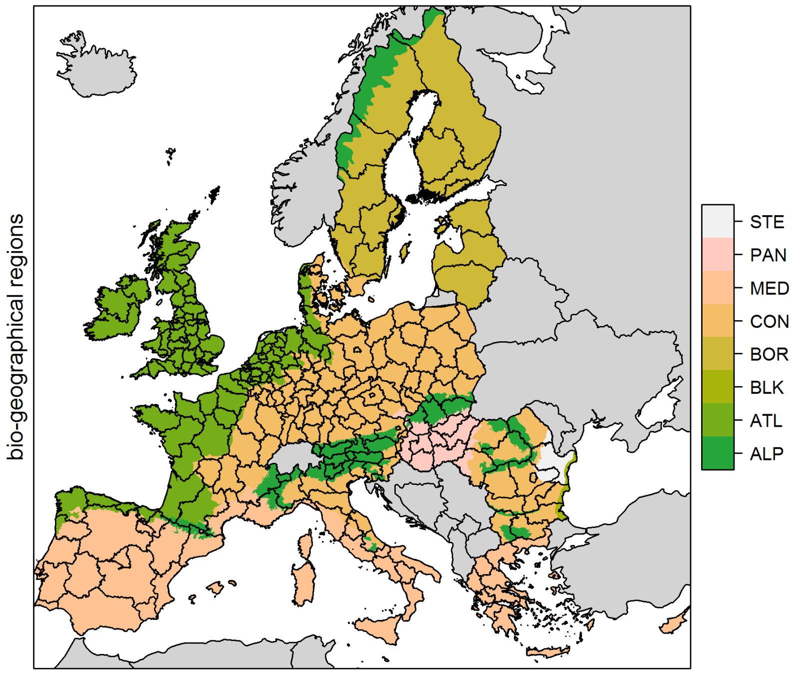

FIGURE 6.4: Bio-geographical regions in EU-27 countries (Alpine (ALP), Atlantic (ATL), Black Sea (BLK), Boreal (BOR), Continental (CON), Mediterranean (MED), Pannonian (PAN) and Steppic (STE) regions). *Source:* authors' computation based on European Environment Agency (2015).

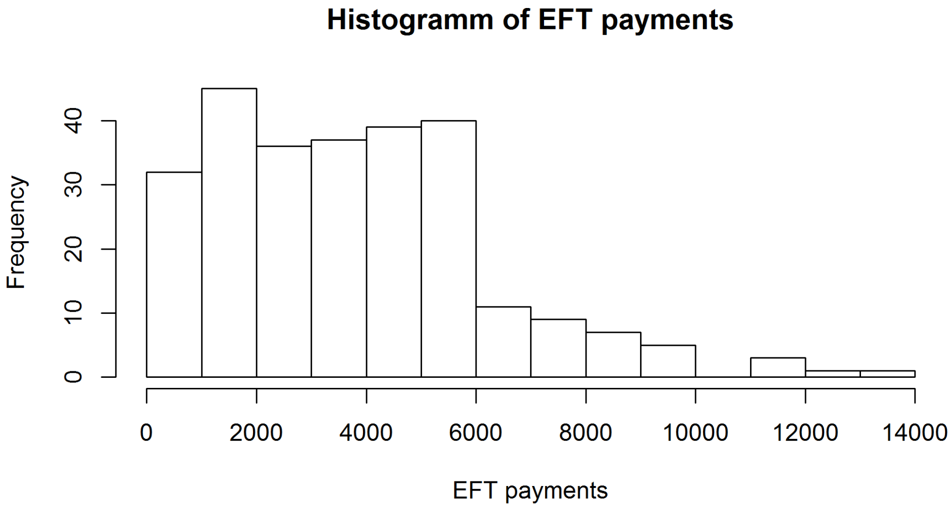

FIGURE 6.5: The frequency distribution plot of simulated EFT payments in €1,000, with mean at dashed line. *Source:* authors' computation based on European Environment Agency (2015).

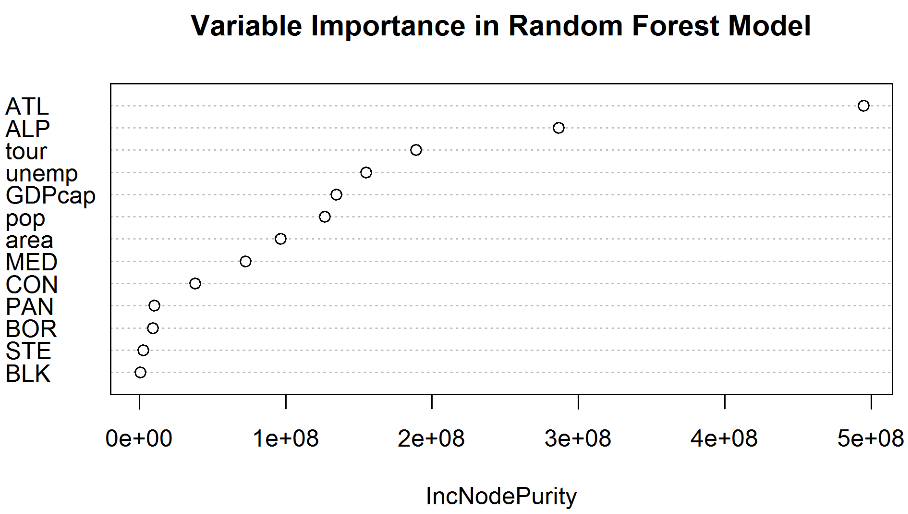

FIGURE 6.6: The variable importance in Random Forest Model with 10,000 trees and a tenfold cross-validation. The x axis displays the average increase in node purity by splitting the variables at the y axis. Variables are Atlantic (ATL), Alpine (ALP), overnight stays in tourist accommodation establishments (tour), unemployment in per cent (unemp), and GDP per capita (GDPcap), populations density (pop), area in km^2 (area), Mediterranean (MED), Continental (CON), Pannonian (PAN), Boreal (BOR), Steppic (STE), and Black Sea (BLK) bio-geographical regions. *Source:* authors' computation based on European Environment Agency (2015) and Eurostat (2016).

Designing a global mechanism for intergovernmental biodiversity financing

This chapter is currently a draft article in preparation for submission to Nature

Droste, N., Farley, J., Ring, I., May, P.H., Ricketts, T. (2017) Designing a global mechanism for intergovernmental biodiversity financing.

Abstract: The Convention on Biological Diversity (CBD) and the Nagoya Protocol display a broad international consensus for biodiversity conservation and an equitable sharing of benefits. The CBD Aichi biodiversity targets show a need for both additional action and enhanced mobilization of financial resources. A proposal of financial burden sharing among states has not yet been developed. We contribute a policy design study for a global scale financial mechanism to support biodiversity conservation through intergovernmental transfers. We develop three design options: ecocentric, socio-ecological and anthropocentric. We analyze the corresponding incentives to reach the Aichi target of terrestrial protected area coverage by 2020. The socio-ecological policy design provides the strongest incentives for states with the largest distance to the Aichi target. Thereby, we provide a new and innovative instrument proposal for global biodiversity financing. We anticipate it to be a starting point for more specific policy dialogues on intergovernmental burden and benefit sharing.

Keywords: biodiversity financing, Convention on Biological Diversity, ecological fiscal transfers

JEL codes: H77, H87, Q57

7.1 Introduction

In order to safeguard human survival on the planet through conservation and sustainable use of biological diversity, the Convention on Biological Diversity (CBD) aims at institutionalizing benefit sharing and appropriate funding mechanisms. While the convention recognizes national sovereignty as a governing principles, it also affirms that the conservation of biodiversity is a "common concern of humankind" (UN, 1992). The parties to the convention agreed upon implementing biodiversity strategies, monitoring, and conservation policies nationally. On the international arena, access and benefit sharing (ABS) mechanisms have further been specified in the Nagoya Protocol. These mechanisms are meant to facilitate 'fair and equitable sharing of benefits' that originate from the utilization of genetic resources and 'appropriate funding' (UN, 2011). In this context benefits are understood in terms of both economic and non-economic values which can be shared between states and between private and state actors (Morgera, 2016; Morgera, Tsioumani, and Buck, 2014). Private benefits may refer to direct use values from bioprospecting and marketization of inputs gained from genetic resource material and information (Convention on Biological Diversity, 2002; Oldham, Hall, and Forero, 2013; Székely and Gaillard, 2007; UN, 2011). Public benefits range from insurance values of safeguarding habitats, ecosystems and life sustaining biospheric systems, over option values of yet unknown future uses, to spill-over benefits that arise from spatial interactions among ecosystems such as the multiple habitats of migratory species (Bartkowski, 2017).

There are 20 strategic targets of the CBD for 2020, the Aichi targets. They are structured in 5 goals, i) mainstreaming biodiversity policies, ii) pressure reduction and sustainable use, iii) safeguarding ecosystems, species and genetic diversity, iv) benefit enhancement, and v) improving implementation (Convention on Biological Diversity, 2010). Most target elements show some but insufficient progress to reach the Aichi targets by 2020, some do not show no significant overall progress, some show movement away from the target, and very few target elements show sufficient progress (CBD, 2014). One of the main causes of insufficient progress is inadequate financing (Balmford et al., 2003; McCarthy et al., 2012; McClanahan and Rankin, 2016; McKinney, 2002; Waldron et al., 2013). Most conservation spending in developed countries comes from domestic sources while developing countries mainly rely on inter- and transnational biodiversity financing (Waldron et al., 2013). The international funding comes through the UN Agencies like the Global Environmental Facility (GEF) who finances CBD related projects and further bilateral agreements (Waldron et al., 2013). The lack of overall progress towards the Aichi targets calls for additional action and innovative financial mechanisms (CBD, 2014). Article 10 of the Nagoya Protocol declares that a global multilateral access and benefit-sharing mechanism (ABS) "to support the conservation of biological diversity" shall be considered by the parties (UN, 2011). The ABS mechanisms are expected to create economic incentives for biodiversity conservation but no direct (financial) obligations arise from the formulation of the article and an corresponding mechanism design has yet to be developed (Morgera, 2016; Morgera, Tsioumani, and Buck, 2014). We approach the search for an ABS innovation guided by a principle of fiscal equivalence (Olson, 1969).

The principle has been developed for the financing of public goods and services. It states that those who benefits from the good in question should also pay for the costs of provision. It is meant to ensure an efficient provision of public goods and services. While private beneficiaries would thus also have to contribute to a corresponding ABS mechanism or fund (Székely and Gaillard, 2007), we will focus on intergovernmental co-financing. Conservation does not just provide national benefits, it also yields transnational public benefits such as insurance values and spill-over benefits (Bartkowski, 2017). In case of spill-over benefits, the principle of fiscal equivalence calls for intergovernmental transfers in order to compensate those who bear the costs of provision (Olson, 1969). A resulting global ecological fiscal transfer (EFT) (Ring, 2008a) mechanism for the benefit sharing across nation states would provide an important and innovative contribution to reaching Aichi targets. This is especially the case since such a mechanism may incentivize nations to supply global benefits of conserving biodiversity through protected areas (Droste et al., 2017c; Farley et al., 2010; Loureiro, 2002; May et al., 2002; Ring, 2008c).

7.2 Developing mechanism designs

Largely unnoticed by the international community, Brazilian states have invented and implemented EFT since the early 1990s. In order to compensate municipalities for the opportunity costs of hosting state and national protected areas on their territory, the state of Paraná implemented a mechanism that distributes a portion of tax revenue according to the the location of protected areas in 1991 (Loureiro, 2002; May et al., 2002; Ring, 2008a,c). Several other Brazilian have subsequently implemented their own EFT schemes such that currently 17 out of 26 Brazilian states have adopted various designs of the instrument (Droste et al., 2017c; May et al., 2002; Ring, 2008c). First impact studies show that the implementation of EFT schemes creates an incentive for the receiving municipalities to increase municipal protected areas (Droste et al., 2017c; Sauquet, Marchand, and Féres, 2014). In recent years EFT have gained international recognition and Portugal has implemented a similar scheme at the national level in 2007 (Santos et al., 2012). Several proposals have been developed for Switzerland, Germany, Poland, France, Indonesia and India and the EU network of protected areas (Borie et al., 2014; Droste et al., 2016, 2017b; Irawan, Tacconi, and Ring, 2014; Köllner, Schelske, and Seidl, 2002; Kumar and Managi, 2009; Ring, 2008b; Schröter-Schlaack et al., 2014). An adaptation to the global level has been proposed (Farley et al., 2010) but has not yet been designed or simulated.

Based on the Brazilian instrument, we propose three design options. The *ecocentric design* would be based on protected area extent per country, irrespective of any socio-economic factors. Each country's environmental indicator, EI, would be calculated as the sum of all protected areas PA weighted with w_k regarding the International Union for Conservation of Nature (IUCN) protected area category k according to their contribution to conservation goals (equation 7.1).

$$\mathrm{EI}_i = \sum_{j=1}^{n} \mathrm{w}_k \mathrm{PA}_{ij} \tag{7.1}$$

The *socio-ecological design* would furthermore take into account the country area and fairness by accounting for relative protected area share per country's area – which is in line with the percentage goal of Aichi target 11 – times a reversed Human Development

Index (HDI), where less developed countries would obtain a relatively larger share of the fund (equation 7.2).

$$\mathrm{EI}_i = \sum_{j=1}^{n} \mathrm{w}_k \frac{\mathrm{PA}_{ij}}{\mathrm{area}_i}(1 - \mathrm{HDI}_i) \tag{7.2}$$

The *anthropocentric design* would extent the socio-ecological design by accounting for population density, which would increase the effect for countries that have both many protected areas and people – which would maximize people's benefits from protected areas (equation 7.3).

$$\mathrm{EI}_i = \sum_{j=1}^{n} \mathrm{w}_k \frac{\mathrm{PA}_{ij}}{\mathrm{area}_i}(1 - \mathrm{HDI}_i)\frac{\mathrm{pop}_i}{\mathrm{area}_i} \tag{7.3}$$

The fund would then be distributed among all L countries according to their EI (equation 7.4).

$$\mathrm{EFT}_i = \mathrm{fund}\frac{\mathrm{EI}_i}{\sum_{l=1}^{L} \mathrm{EI}_l} \tag{7.4}$$

For details on the calculations beyond the general design options see the method section.

7.3 Resulting financial flows and incentives

For the mechanisms we have computed the protected area extent and relative shares based on United Nations Environment Programme (UNEP) Protected Planet data for all IUCN categorized protected areas and Global Administrative Areas country shapefiles (GADM, 2016; UNEP, 2016). For the spatial analysis we followed the UNEP guide, for details see the method section link. The fairness indicator is based on United Nations Development Programme (UNDP) HDI data (UNEP, 2017). Population data has been provided by the World Bank (World Bank, 2017). We have simulated the resulting EFT per country for each of the design options for all national CBD parties (first column, Figure 7.1) – which includes all UN Member States except the USA. Then we computed the resulting incentives as a change in EFT flows to a country if it unilaterally increases its protected areas by 1 per cent of its area, ceteris paribus (second column, Figure 7.1). We also calculated how much of an incentive would result per square kilometer (third column, Figure 7.1).

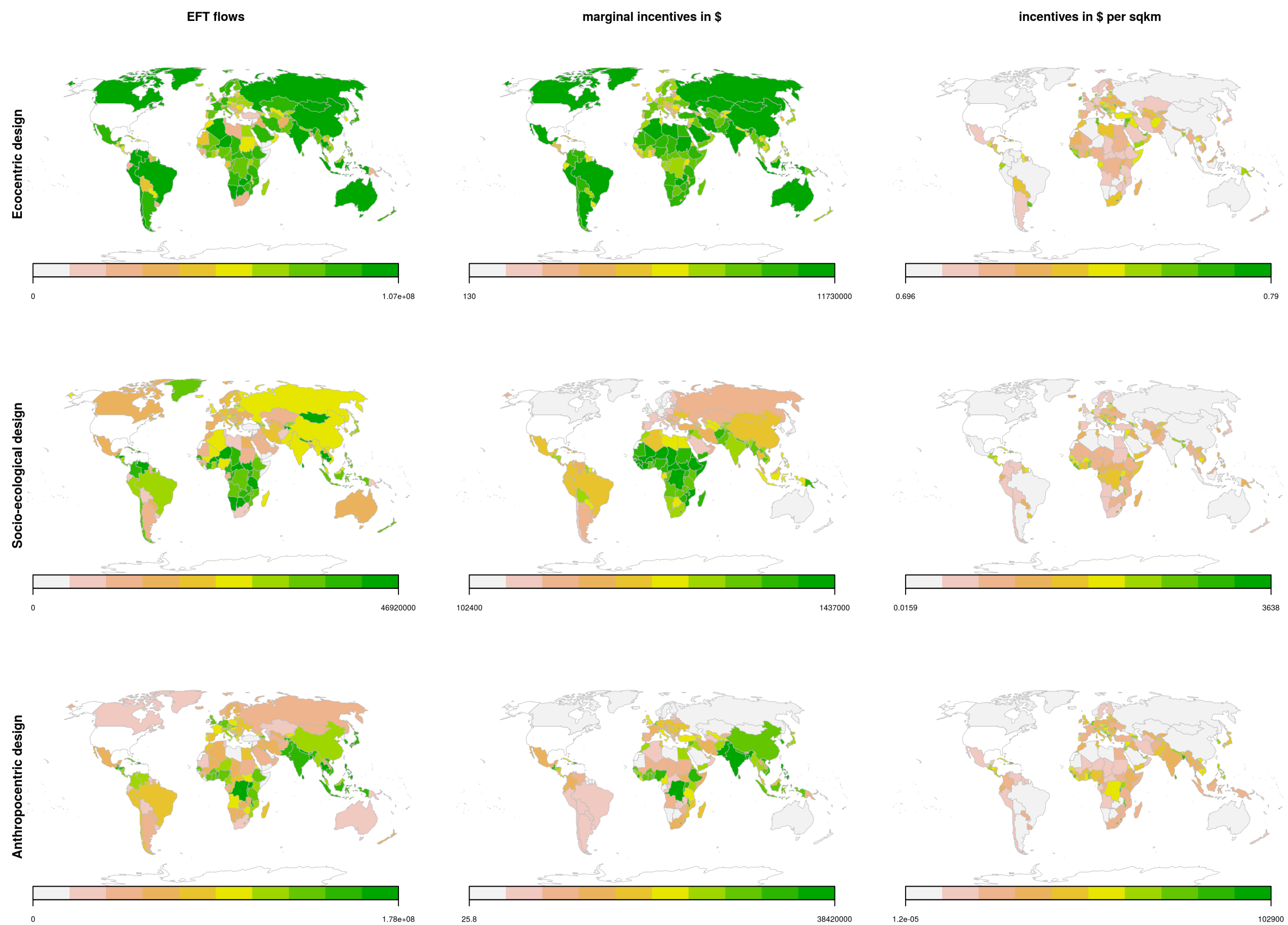

FIGURE 7.1: Global maps of different EFT designs and the resulting incentives. Incentives are computed as the marginal and per sqkm change in EFT flows for a unilateral protected area increase of 1 per cent total area per country, ceteris paribus. The countries are color coded in deciles and the legends display an equal spacing per quantiles while the actual values are distributed differently among deciles. Maps have a Robinson projection. *Source:* authors' elaboration.

7.4 Design choice based on Aichi target 11

In order to assess which instrument design is the best choice we evaluate how far countries are from reaching Aichi target 11 which states that by 2020 17 per cent of all terrestrial land shall be protected (Figure 7.2).

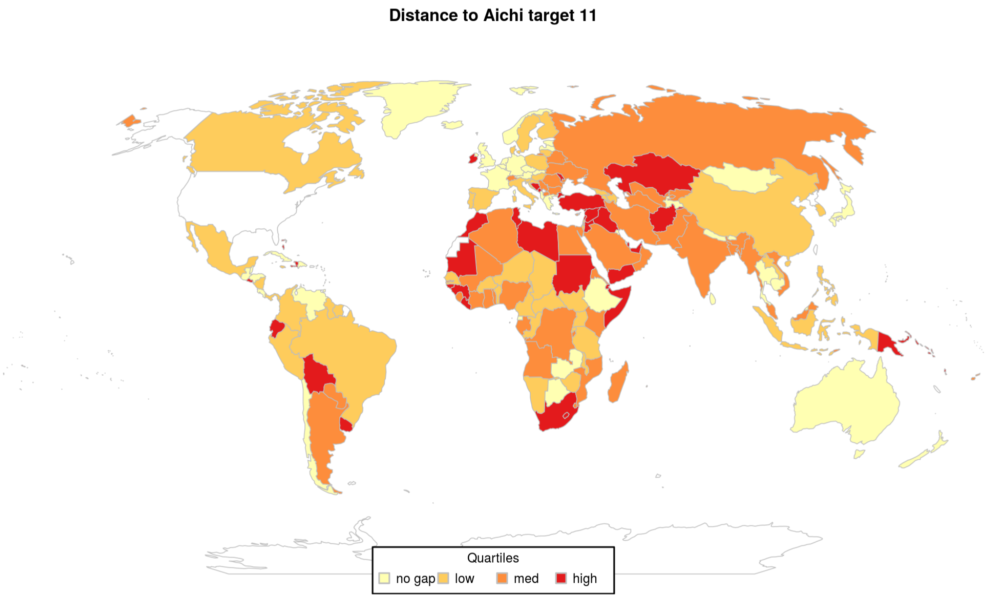

FIGURE 7.2: Global map of countries' distances to fulfill Aichi target 11 of 17% terrestrial protected areas by 2020, calculated as 17 minus countries current protected area share in percent. Only IUCN categorized protected areas are considered. The countries are grouped in quartiles. Quartile colors are lightyellow for a distance of less than 0 up to 1.14, lightorange for up to 8.91, darkorange for up to 15.10 , red for up to 17.00. Non-CBD countries are white. The map has a Robinson projection. *Source:* authors' elaboration.

We grouped the countries' distances to Aichi target 11 by quartiles and computed the distribution of both marginal and per square kilometer incentives per quartiles for each of the three design options. The design choice is based on the following consideration. The strongest incentive should go to those countries that are the farthest from reaching the Aichi target. They are the ones that need to increase protected area share the most and thus should be incentivized most. Figure 7.3 provides boxplots of incentives per design for both marginal and per square kilometer incentives. The socio-ecological design consistently provides the highest mean incentive for the quartile of countries that have the largest distance to reaching Aichi target 11 and thus ranks first. The social design ranks second and the ecological design comes last.

7.5 Design choice implications

Distributing a biodiversity fund according to the location of protected areas compensates for past efforts and sets incentives for creating additional protected areas since they become a source of income (Farley et al., 2010; May et al., 2002). We contribute the

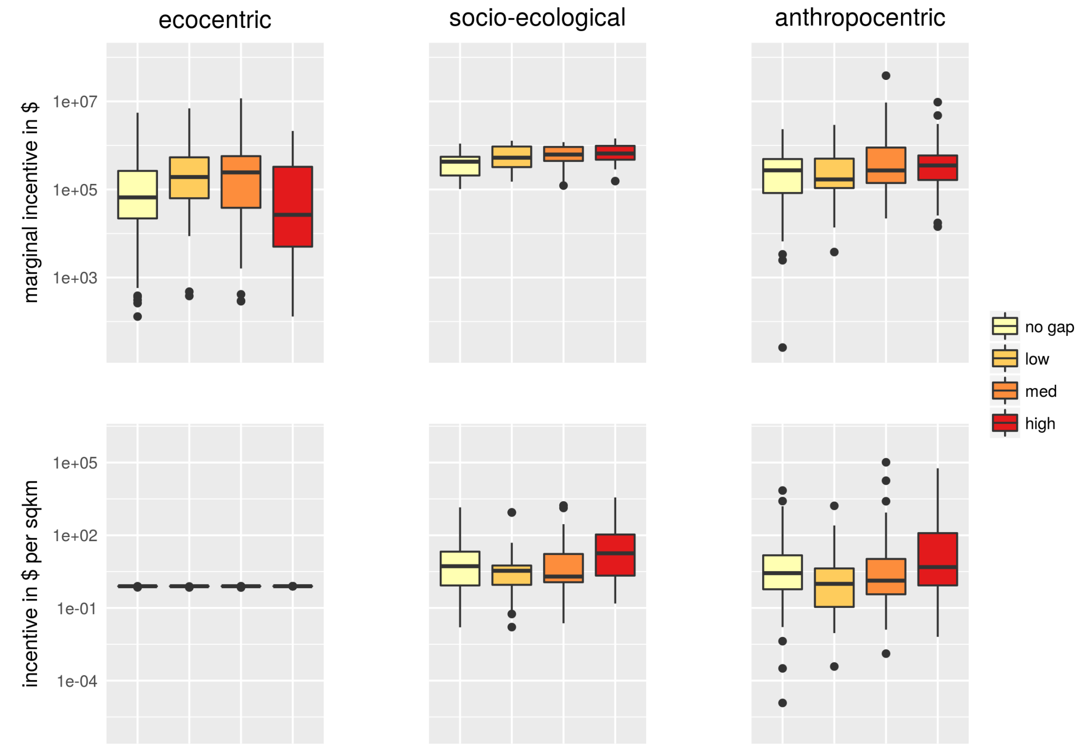

FIGURE 7.3: Per quartile distribution boxplots of incentives for the design options. First row indicates the marginal incentive in terms of an EFT change from a unilateral per country increase of its protected area share by one per cent. The second row indicates the incentives as a per square kilometer EFT change. Countries are categorized into quartiles according to countries' distances to fulfill the Aichi target of 17% terrestrial protected areas by 2020. The quartiles are "no gap" for a distance of less 0 up to 1.8, "low" for up to 8.9, "med" for up to 15.2 , and "high" for up to 17. The Y-axes of the are log transformed and equal across the three design options per row. *Source:* authors' elaboration.

first policy design study on a global intergovernmental fiscal transfer scheme to support biodiversity conservation. The socio-ecological design option allocates the fund such that those countries showing the least progress towards reaching a 17 per cent protected area share by 2020 receive the strongest financial incentive to designate additional protected areas.

hereby we would expect these countries to have the highest probability to respond to an implementation of the global EFT with increasing their protected area share. The instrument can thus help to reach Aichi target 11. Although Aichi target 11 is one of the few targets that shows sufficient progress recent contributions argue that humanity needs to protect half the Earth in order to safeguard biodiversity (Dinerstein et al., 2017; Wilson, 2016). We would thus expect that Aichi target 11 will be increased after 2020. The design choice would still be the same if the distance to 30 or 50 percent was the underlying criterion. Important possible future extensions include biodiversity targeting, directing flows where biodiversity is highest or most threatened, and the inclusion of marine protected areas. But even in its most basic form the instrument would contribute to other Aichi targets than just target 11. It would help to mainstream biodiversity (target 1) into fiscal planning and other policy arenas (target 2) for its intergovernmental fiscal nature. It is an instrument that provides positive incentives for biodiversity conservation (target 3). It would help to reduce the loss of habitats (target 5) and since the less strict protected area categories are taken into account it would also help to ensure sustainable land management (target 7). It also links well with the Nagoya Protocol on ABS (target 16). It would require financial resources to set up an EFT fund at the global level and is thus in line with target 20 on increasing biodiversity financing. Since the largest sums would flow mainly to low income, small island states and to a large extent to African countries, the socio-ecological design does support a fair and equitable distribution of benefits among nations.

The proposed instrument is thus well aligned with the current regime complex for biodiversity protection (Buck and Hamilton, 2011; Raustiala and Victor, 2004; Rosendal, 2006). It furthermore has the benefit of being implemented in similar forms among Brazilian states and in Portugal, such that actual experiences can be further explored and analyzed regarding design principles and outcomes. The main value added by the current proposal, however, consists in the upscaling of an existing instrument for biodiversity conservation to the global level. As such it fills a gap on how ABS mechanisms can be implemented and provides an innovative contribution to the current debates. We would expect that our proposal may serve as a starting point for a more specific science-policy dialogue on benefit and burden sharing of biodiversity conservation between the CBD, the Intergovernmental Platform on Biodiversity and Ecosystem Services, and the broader community.

7.6 Methods

This section includes methodological details on: a) the calculation of protected areas per country, b) the construction of a dataset including socio-economic control variables, c) the computation of distributive patterns per design option, d) the computation of distance to Aichi target 11, e) the computation of assessment criteria for design option selection. All source code in both python and R can be found at a personal github repository at: [link to be inserted] such that the results of the analysis are entirely reproducible.

(A) Calculation of protected areas per country

We downloaded the UNEP-WCMC global data set of protected areas from `www.protectedplanet.net` (version May 2017) as a .gdb file. We used ArcGIS (version 10.4) to compute the share of IUCN protected area categories per country with the following algorithm (based on adapted form of the UNEP-WCMC method): we repaired geometry features for both point and polygon data were repaired; protected areas with statuses 'Not Reported' and 'Proposed' were omitted. We excluded protected areas that are classified as 100% marine, and point data that had no reported area. The point data was reprojected to World Equidistant Cylindrical coordinate reference system (CRS)

(ESRI:54002), points were buffered such that the buffer area matched the reported area and reprojected to World Behrmann CRS (ESRI:54017); polygon data was directly reprojected to World Behrmann CRS; reprojected polygon and buffered point data were merged into a single .gdb. Spatial data on country outlines was obtained from Global Administrative Areas database (`www.gadm.org`) and reprojected to world Behrmann CRS. For each of the IUCN protected area categories (Ia, Ib, II, III, IV, V, and VI) the corresponding protected areas we dissolved, repaired and iteratively erased from overlaps with former category areas, repaired again, and the country intersection with protected areas was tabulated. Finally, the polygon attributes were exported as a .csv file.

(B) Construction of a dataset including socio-economic control variables

The per country IUCN category protected area data was loaded into **R** (version 3.4.1). Only countries party to the CBD were selected (including Greenland). UNDP data on HDI was added from `http://hdr.undp.org/en/data` (2015 data, published 2017). Per country data on population was downloaded from the Worldbank Database through the "***WDI***" package. All these datasets were joined into a single dataframe;

(C) Computation of distributive patterns per design option

We used weights for IUCN protected area categories to account for their different contribution to conservation goals based on an adaptation from weights in the Brazilian EFT scheme: w =(Ia= 1, Ib= 0.9, II= 0.8, III= 0.7, IV= 0.5, V= 0.3, and VI= 0.1). The design option payments per country were calculated according to formulas 7.1 – 7.4 in the main text.

(D) Computation of distance to Aichi target 11

The distance, D, was calculated as $D_i = 17 - 100\frac{\sum_{j=1}^{J} \mathrm{PA}_{ij}}{\mathrm{area}_i}$, for all J protected areas in country i. Countries were then grouped in quartiles according to D.

(E) Computation of assessment criteria for design option selection

The marginal incentives per countries were computed as the additional transfer for a unilateral increase of a 1 per cent protected area increase with a probability distribution over IUCN protected area categories corresponding to global average probabilities of the categories. The per square kilometer incentives were calculated as marginal incentive per square kilometer country territory. Both the marginal and per square kilometer incentives were plotted in box plots according to the quartiles of distance to Aichi target 11. The choice was made based upon which design provides the strongest incentives for those countries farthest from Aichi target 11.

Part V

Conclusions

Lessons Learned and Value Added

A short primer on policy advice for Ecological Fiscal Transfers

Against the backdrop of an ongoing loss in biodiversity and a lack of success in halting this loss, this dissertation analyzes a promising innovative fiscal instrument that may help nature conservation efforts. EFT tie a share of tax revenue distribution among lower level governments to the existence of PA – which thus become a source of income. It may thus incentivize the designation of additional PA – so far a commonly held belief which had yet lacked systematic analyses.

Three interlinked questions about the instrument were therefore answered in this dissertation:

1. Does the implemenation of EFT lead to an increase in PA?
2. Which institutional characteristics of the EFT schemes determine the outcome?
3. What policy advice can be drawn from the empirical analyses and quantitative modelling for the design and adaptation of EFT schemes?

As a starting point for the subsequent analyses, this dissertation developed a theoretical microeconomic public finance model of how the performance-oriented transfer design creates financial incentives to increase the amount of PA. Testable hypotheses were derived from modeling the effect of a change in relative costs in both a unilateral decision making context of a single local government and a bi-directional setting where two local municipalities compete for the funds available through EFT. The short-listed hypotheses to be tested econometrically were (in a condensed form): a) EFT increase PA, and b) EFT has an equalizing effect.

In order to answer the first two questions and to assess the first hypothesis, existing EFT schemes were analyzed with econometric techniques to assess effects on the designation of additional PA. The *Brazilian case study* shows through a microeconometric panel data analysis that there is a significantly higher share of total PA in states with EFT. While this may be both a cause for or an effect of the EFT, the identification strategy shows that the designation rate of municipal PA over time increases on average after the introduction of EFT. This increase in municipal PA that can thus be attributed to the implementation of EFT.

The *Portugues case study* furthermore displays the importance of institutional requirements for such a decentralizing effect through the usage of a Bayesian structural time series approach. A local government competency for designating protected areas is not just a prerequisite, it may also enhance the effect of an EFT if competencies are widened. The first two research questions can thus be answered jointly: an EFT implementation can lead to an additional increase in *decentral* PA – given that the lower government levels have the legal competency to designate local PA.

The next three case studies provide evidence based policy designs to exemplify what policy advice can be drawn for the adaptation of EFT schemes to higher governmental levels. The *German case study* shows how an EFT mechanism can be integrated within federal-state fiscal relations. It provides an institutional analysis of the German fiscal equalization scheme, gives econometric evidence for a structural condition for above-average fiscal needs for nature conservation among states, and simulates the marginal fiscal transfers resulting from an integration of ecological indicators. The study thus exemplifies how a combined institutional, empirical and policy design analysis can be applied to upscale the originally municipal EFT mechanism to inter-state fiscal relations.

The *EU case study* simulated an EFT scheme among EU regions based on Natura 2000 sites. The spatially explicit distributional effects were assessed through a regression tree model on socio-economic characteristics of beneficiaries of the proposed policy design. The largest transfers would flow to mountainous, touristically less developed and economically poorer regions. The study does thus not only provide evidence for the third hypothesis by showing that EFT may indeed have an equalizing effect. It also showed that this distributional effect fits the cohesion policy goal of the EU Regional Development fund - which has thus been identified as a suitable fund for implementing an EU EFT scheme.

The policy design study on an *international EFT mechanism* developed three different design options: ecocentric, socio-ecological, and anthropocentric. The designs were simulated and assessed in terms of resulting financial incentives for designating additional PA. Based on the assumption that the incentives should be strongest where most effort is required to reach internationally agreed upon biodiversity targets, the socio-ecological design showed the greatest promise. The study contributes an evidence based design study regarding adaptation of a decentral mechanism up to the global scale and may spure a science-policy dialogue on global, intergovernmental burden sharing for biodiversity finance. Again, the main beneficiaries under the socio-ecological design are poorer small island and African states.

Through these studies the disseration contributed i) a first systematic empirical assessment of the effect EFT can have on the designation of decentral PA through two econometric studies on existing schemes – which can be positive if the necessary competencies at the receiving level are given. The dissertation furthermore provided ii) evidence-based policy proposals for state-federal fiscal relations, the EU and the CBD through assessing both conservation efforts, and the distributional and incentive effects inherent in the proposed and simulated mechanisms. This allowed for providing evidence of a structural above average fiscal need for conservation in Germany. It showed the suitability of EFT with cohesion policy goals in the case of the EU study. In the case of the CBD study it allowed for choosing a mechanism design that creates the strongest incentives where there is the greates lack of conservation action. The dissertation has thus not just provided empirical evidence for the contribution of EFT to conservation policy goals but also provided evidence-based assessments of three potential adaptations of the instrument to federal, supra- and international settings.

Given the scale and velocity of biodiversity loss societal responses are required in order to mitigate some harmful outcomes. Yet, what this dissertation has not directly dealt with is that if funds are directed towards biodiversity financing these monies are not necessarily available for other societal needs. This is an apparent trade-off. However, fiscal transfers are tied to the existence of PA (the performance-based element) but the received revenue can be spent on whatever needs the respective administration sees fit (the general purpose element). Thus, by design, EFT minimize the trade-offs by setting incentives through the performance-based transfers while other societal needs can still be paid for by the transfers. The equalizing characteristic of EFT indicates a greater supply of PA in economically weaker municipalities or regions and the funds are hence directed where there is less tax income from economic activity. Yet, the design of the instruments determines whether such patterns actually equalize the distribution of revenue. EFT schemes thus show potential for synergies between biodiversity conservation and reduction of inequality.

Summarizing, three general policy recommendations can be drawn:

1. If an EFT scheme is to enhance PA it will be important to ensure that there are sufficient conservation compentencies at the addressed government level.

2. If an EFT scheme is adapted to a new (inter-)governmental setting institutional analyses and quantitative simulations may help to select policy design features.

3. If an EFT is to reduce inequalities and equalize (tax) revenue the design of the scheme will need to take such double purpose into account.

These recommendations – while reductionistic – have wider implications. They call for careful assessments and designs of instruments that have a potential to contribute to conservation policy goals. Moreover, they call for policy designs that are based on (trans-)disciplinary scientific evidence. They also imply the potential of mainstreaming and upscaling a decentral policy innovation, namely the integration of ecological indicators in intergovernmental fiscal relations. By adapting and tailor-making the instrument to fit different institutional settings, EFT have the potential to incentivize nature conservation policies at various government levels while simultaneously reducing inequality.

Part VI

Appendix

Slutsky Identity

The total effect of the relative price change can be decomposed with the Slutsky equation into a substitution effect and an income effect (Varian, 2010, chapter 8). Consider the change in demand for X as a function of price p_x and available budget M, then the total effect of such a price change in relation the demand for X can be written as a sum of Slutsky substitution effect, ΔX^s, and Slutsky income effect, ΔX^n, with $\overline{p_x}$ as the old price, p_x as new price, and M^s as Slutsky adjusted income:

$$\Delta X = \Delta X^s + \Delta X^n \tag{A.1}$$

$$X(p_x, M) - X(\overline{p_x}, M) \equiv [X(p_x, M^s) - X(\overline{p_x}, M)] + [X(p_x, M) - X(p_x, M^s)] \tag{A.2}$$

Note, that this is an identity since first and fourth right hand terms cancel out, both sides are identical, and it is thus true for all values of p, p', and M (Varian, 2010, p. 143).

The shortest, and one of the most intuitve proofs of the Slutsky equation has been provided by Cook (1972). Nevertheless, I will follow (Varian, 2010, appendix to chapter 8) in deriving the Slutsky equation in order to provide the correct effect of a price change in p_x on X (instead on Y as in Cook). But it is basically the same approach.

Let the original consumption bundle be denoted as $(\overline{X}, \overline{Y})$ at their prices $(p_{\overline{x}}, p_{\overline{y}})$, and income $\overline{M}$. Consider a price change to (p_x, p_y) while we adjust money income to M^s, such that the old consumption bundle can still be afforded. Formulating a Slutsky demand function for X^s, we state

$$X^s(p_x, p_y, \overline{X}, \overline{Y}) \equiv X(p_x, p_y, p_x\overline{X}, p_y\overline{Y}) \tag{A.3}$$

which tells us what the consumer would demand facing the new prices (p_x, p_y), and having income $(M^s = p_x\overline{X}, p_y\overline{Y})$. Differentiating this equation with respect to p_x, employing the chain rule, we get

$$\frac{\delta X^s(p_x, p_y, \overline{X}, \overline{Y})}{\delta p_x} = \frac{\delta X(p_x, p_y, M^s)}{\delta p_x} + \frac{\delta X(p_x, p_y, M^s)}{\delta M^s}\overline{X}\,. \tag{A.4}$$

We can rearrange to

$$\frac{\delta X(p_x, p_y, M^s)}{\delta p_x} = \frac{\delta X^s(p_x, p_y, \overline{X}, \overline{Y})}{\delta p_x} - \frac{\delta X(p_x, p_y, M^s)}{\delta M^s}\overline{X}\,. \tag{A.5}$$

which is a derivative form of the Slutsky equation, telling us that the total effect of a price change is composed of a substitution effect (adjusting income to allow for consumption of $(\overline{X}, \overline{Y})$), and an income effect (Varian, 2010, appendix to chapter 8).

Source Code & Data

- Empirical Analyses
  - `https://github.com/NilsDroste/EFT-BR`
  - `https://github.com/NilsDroste/EFT-PT`
- Policy Design Studies
  - `https://github.com/NilsDroste/EFT-DE`
  - `https://github.com/NilsDroste/EFT-EU`
  - `...`

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