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The impact of agri-environmental policy and infrastructure on wildlife and land prices

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Contents

Chapter 1

Introduction

Contemporary rates of decline in biodiversity are dramatic (Barnosky et al., 2011; Newbold et al., 2015). Key drivers of biodiversity loss have largely been anthropogenic, such as land use changes (soil sealing through urbanization and infrastructure, intensive agriculture), fragmentation of habitats, air and water pollution, and climate change (Kok et al., 2018; Pereira et al., 2010). Given its commitments to the Aichi Biodiversity Targets, in 2011 the European Union (EU) adopted a new strategy to protect biodiversity including six specific targets to be reached by 2020 (EU, 2011). Two key legal documents that have shaped European environmental policy are the birds directive (Council Directive 79/409/EEC of 2 April 1979 on the conservation of wild birds) and the habitats directive (Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora). The diverse objectives and strategies to reduce biodiversity loss are operationalized using command and control approaches as well as economic instruments.

Despite the efforts to reduce the environmental impact of development, the 2015 mid-term review of the EU's biodiversity strategy has revealed slow or no progress towards 2020 biodiversity targests. A recent report by the European Environmental Agency (EEA) points out that land use is changing faster, including changes from agricultural land into artificial surfaces (e.g. urban areas and infrastructure). Agricultural land in the EU decreases at a rate of 1000 km^2 per year, on average. Forest area has remained stable, but intensification in forest use has been noticed, which may lead to declining habitat quality of forest ecosystems (EEA, 2017).

While the theory on the economics of environmental policy is well established (Phaneuf & Requate, 2016), the practical implementation remains challenging. Payments for environmental services (PES) are a classical example that can only work effectively if there is little to no information asymmetry between the regulator and the individual targeted by the policy. In the EU, PES schemes have been adopted in the second pillar of the Common Agricultural Policy (CAP) in order to reduce the environmental impacts of agriculture. A key issue, which is studied in the first paper of this dissertation, is the effectiveness of PES (Börner et al., 2017; García-Amado, Pérez, Escutia, García, & Mejía, 2011; Wunder, 2007). As a recent review by Börner et al. (2017) revealed, information asymmetries make many PES schemes vulnerable to ineffectiveness due to adverse selection. This results in public payments that do not induce changes in farmers' behavior, and may therefore fail their environmental and other goals.

A second issue are possible side-effects of conservation policy. Environmental policy does not unfold its effects within a closed-off system, therefore undesirable or unanticipated feedbacks to markets are likely and difficult to avoid. It is therefore important to not only study the goal-attainment of a policy (Vedung, 1997), but also any undesired side-effects. The European Natura 2000 network of protected areas is a prime example to study these effects, as it follows an integrated approach to conservation in that, compared to other conservation concepts, it does not exclude economic activity. Farming, particularly extensive, low impact farming, is seen as an essential ingredient to effective conservation of certain habitats and species of community interest (Halada, Evans, Romão, & Petersen, 2011). The key challenge in the integration of farming and conservation is the balancing between conservation measures leading to production restrictions, and financial compensation. Our second paper studies the impact of Natura 2000 designation on farmland on land prices using the generalized propensity score method (Imbens & Hirano, 2004).

The third paper departs from the agricultural context and studies the impact of highway construction on wildlife species. From an ecological perspective, the literature gives several rationales on how highway construction can affect wildlife populations, in particular through habitat destruction and degradation, and through landscape fragmentation. We further elaborate the role which environmental impact assessments have played, they way they were conducted, and their impact they had on certain indicator species given an increase in density of the highway system in Austria.

With this dissertation, I hope to improve the knowledge base for the future design of environmental policy in order to contribute to the achievement of the 2020 biodiversity goals. Environmental policy can only be effective if goals are clearly defined and measurable, if the data are available and monitoring is in place, and if not only goals but also side effects can be evaluated reliably. This requires the combination of many data sources and evaluation approaches. The chapters of this dissertation present approaches to policy and side effect evaluation using a diverse array of econometric methods, including dynamic panel data econometrics, treatment evaluation based on the generalized propensity score, and latent class models. Moreover, I combine and use readily (and publicly) available data to relate the policy to the potential outcome.

1.1 Introduction to the topics

The topics we study in this dissertation are diverse, but they are related through their goal to make the design of environmental policy more effective. In the next sections, we give a short overview of the topics studied in this dissertation and explain how each paper tries to contribute to the solution of a specific issue.

1.1.1 Environmental policy and agriculture

Environmental policy has infiltrated agricultural policy making worldwide. In Europe, the Common Agricultural Policy (CAP) of the EU has introduced environmental requirements for farmers through both pillars, albeit with varying focus according to national and provincial preferences. These environmental requirements are connected to CAP payments shown in Figure 1.1. After the Fischler-Reform in 2003, farmers who received first pillar payments were required to respect, among other regulations, additional environmental constraints on their production, due to a regulation named *cross compliance*. More recently, the greening requirement after 2014 has placed additional constraints on farmers' production, including measures for the conservation of

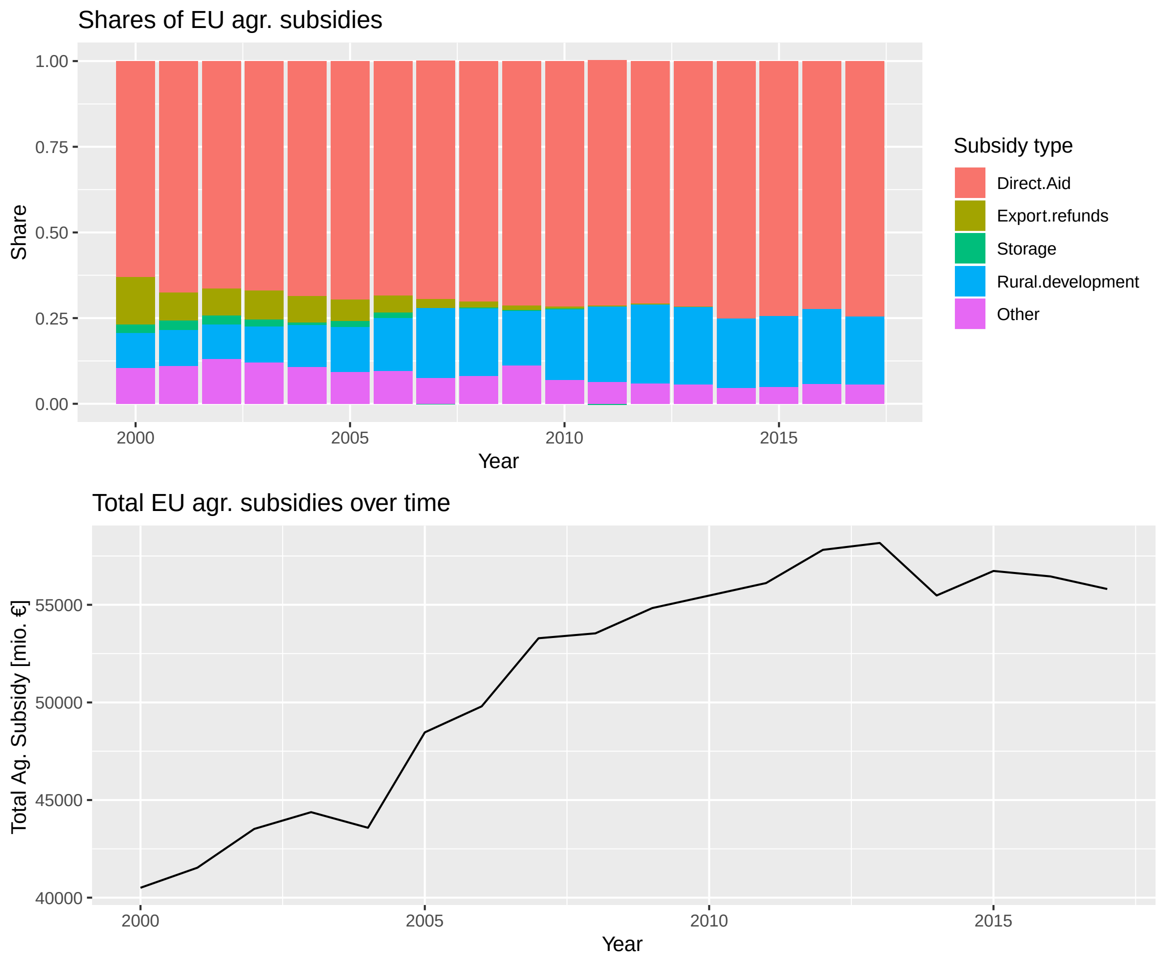

Figure 1.1: Shares and total expenditure of the EU on agricultural subsidies

grassland and the establishment of ecological focus areas.

A second key factor in agri-environmental policy are agri-environmental programs co-funded through the second pillar of the CAP (the European Agricultural Fund for Rural Development, EAFRD) and national budgets. These include financial compensations for voluntary commitments of farmers to reduce their environmental impact, including additional restrictions in fertilizer and pesticide application, habitat-enhancing landscape features (e.g. hedges, grassy margins, single trees), restrictions on crop rotations, and others. These programs are developed by each member state separately, and in some countries (e.g. Germany) even at the subnational (state) level.

Third, through EUs commitment to habitat and species conservation by establishing the network of Natura 2000 protected sites, farmers who own land on designated sites may face additional productivity constraints. Some EU member states, including parts of Germany, offer compensation payments for affected farmers.

A lot of research has been conducted to better understand the impacts of agri-environmental policies in terms of additionality, spill-over effects including leakage, and targeting (Engel, 2016). Payments can be distributed based on activities (the most common case) or based on results. A key feature in PES is the distribution of information. The design of (agri-) environmental policies typically comes with asymmetric information, in that the regulator (i.e. the ministry of agriculture of a given country) only has limited information about the environmental conditions and the biodiversity potential of a given farm (Ferraro, 2008). That puts the regulator at a significant disadvantage for policy design, as he (1) needs to formulate policies broad enough to apply to a wide array of contexts, and (2) he might rely on information provided by local actors when designing the policy.

In our first paper, we study the Austrian agri-environmental program ÖPUL. This program combines a wide variety of more general and more specific measures that are designed to improve farmland biodiversity. Payments are based on participation in a specific measure rather than on outcome. In principle, all farmers are eligible to participate in any measures of their choosing, which may lead to the inefficiency problems outlined above. We argue that when payments based on outcomes are impractical, regional targeting of agri-environmental measures based on observable regional characteristics of both farming and habitat conditions for target species can help to reduce losses from asymmetric information and adverse selection. Regional targeting, as opposed to farm-level outcome-based renumeration, could also be a more useful alternative as indicator species may only be observable at the regional level rather than the farm-level. In our empirical application, we use a latent class framework to disentangle those regions where farmers provide environmental benefits for a given species from those who don't.

1.1.2 The economics of land prices

Prices and rents of farmland are important indicators of the profitability of farms. But farmland prices may not only be a reflection the pure market value of the crops being grown a parcel of land, but are often shaped significantly by policy. Many scholars have studied the effect of various policies on farmland prices (Ciaian, Kancs, & Swinnen, 2012; Ciaian, Kancs, & Swinnen, 2014; Feichtinger & Salhofer, 2013, 2016; Floyd, 1965; Gardner, Moss, & Schmitz, 2003; Goodwin, Mishra, & Ortalo-Magné, 2003; Kilian, Antón, Salhofer, & Röder, 2012; Klaiber, Salhofer, & Thompson, 2017; Michalek, Ciaian, & Kancs, 2014). While land prices are generally thought of to be driven by supply and demand for land, which is heavily influenced by the marginal value of production, some characteristics make land prices particularly sensitive to policies. First, the supply of land is limited, which constrains the expansion as a reaction to demand changes. Second, the marginal productivity of land is often influenced by policies such as different forms of land use rights, zoning, subsidies and taxes. From the farmer's perspective, land prices are thought of to be determined by the expected net present value of net profits. As Goodwin et al. (2003) explain, the effect of policies on land prices may be difficult to measure in practice, and models based on OLS may produce biased results due to endogeneity issues.

A series of papers has produced a theoretical framework of the impact of agricultural policy on farmland prices in Europe (Ciaian et al., 2012; Ciaian et al., 2014). The authors have shown that prices are strongly related to decoupled direct payments, and to the distribution of entitlements for first pillar CAP payments. They also show that the effect of payments linked to additional production constraints, such as cross compliance or agri-environmental programs, may be ambiguous. In our second paper, we study the special case of Natura 2000 designation on farmland in Germany. Natura 2000 designation comes with constraints on farm production that vary by German states. Compared to voluntary agri-environmental programs, Natura 2000 related farming constraints are often mandatory. However, as designation of sites is not a strictly top-down process, farmers may still be able to influence the type of land that is subject to Natura 2000.

In effect, site selection could be non-random, but influenced by many (including political) factors. In an empirical framework, self-selection of low-productivity areas into Natura 2000 could be a problem for the estimation of the effect of site designation on land rental prices. This may lead to substantial overlap issues when comparing land rental prices between Natura 2000 and non-Natura 2000 farms. To address this issue, we employ the generalized propensity score method proposed by Imbens and Hirano (2004).

1.1.3 Environmental policy and infrastructure

Besides agriculture, environmental policy has also strongly influenced the development of large-scale infrastructure. Most countries worldwide have adopted policies to conduct ex-ante evaluations of the environmental impact that an individual project may have. These evaluations are known as Environmental Impact Assessments (EIAs), and they generally include assessments of the expected impact of a project on air quality, water quality, soil, as well as flora and fauna. Besides their effect on the environmental impact of a project, they also serve as documentation of the political processes that lead to specific decisions during the project planning, construction, and operation phases, and are supposed to increase transparency of decision making.

There is a rich literature studying the diverse aspects of EIAs, from study design to implementation to compensation measures and monitoring. Nevertheless, EIAs are often political advocacy documents rather that scientific studies, and many have been published with questionable methods and conclusions. In the context of road construction, EIA quality has been found to be generally poor (Jaeger, 2015). Scholars have particularly criticized missing clarity in the methods used to detect endangered species, the lack of consideration of fragmentation and barrier effects, the descriptive rather than analytical and predictiven nature of most EIAs, rare consideration of indirect impacts, and the focus on the local scale without assessment of habitat fragmentation (Jaeger, 2015, p. 33).

From a European perspective, the TEN-T network of transport infrastructure is an important investment priority that trades off environmental quality with socioeconomic development. The 2011 White Paper on transport by the European Commission sets out the key strategies pursued by the EU (Commission, 2011, p. 5), pointing out that "transport has to use less and cleaner energy, better exploit a modern infrastructure and reduce its negative impact on the environment and key natural assets like water, land, and ecosystems". However, the European Environmental Agency (EEA) has criticized that the relevant EU legislative documents only mention environmental goals in the preamble, and that specific impacts on land would be subject to national planning processes (EEA, 2016). As a result, environmental protection may not have had a high priority in the planning and construction of transportation infrastructure. It is therefore important to improve the knowledge base regarding the environmental impact of highway construction. Our third paper presents an Austrian case study on the effect of highways on important game species over a period of 48 years.

1.2 Theoretical contributions

All of our papers are empirical at their core, however, they also provide some theoretical insights. In the first paper, which studies the impact of agri-environmental policies on wildlife populations, we try to clarify the link between policy design and environmental impact. As reviewed by Uthes and Matzdorf (2012) most studies on agri-environmental programs (AEPs) focus on the pure ecological effect. Other studies focus on farmer farmer adoption and on farm-business related characteristics (e.g. additional costs). But the two sides along the path from policy design to the ecological effect remain largely disconnected in the literature. In addition, AEPs often suffer from adverse selection, particularly when participation is voluntary. By linking the farmers' profit function to an ecological model, we show that only one out of three types of farmer will potentially change habitat conditions for wild animals. The added complexity of the ecological response of a wildlife species leads to additional difficulties in the design of AEPs. Our theoretical model suggests that a substantial share of funds allocated to AEPs may actually not contribute to reaching intended biodiversity goals.

The second paper studies the effect of Natura 2000 conservation policy on land prices in Germany. While the theory on farmland land prices is well established, it often neglects the effects of zoning and other landmarket rigidities. In the theoretical section of this paper, we first explain how Natura 2000 zoning policy will affect the individual farm, and then decompose the district average effect into farms with land on Natura 2000 land and farms without such land. In theory, two effects my change the land price for non-Natura 2000 farms, the first is zoning (i.e. making non-Natura 2000 land more scarce), and the second is funding diversion (i.e. moving public funding from non-Natura 2000 farmers to Natura 2000 farmers). In practice, only the first of these effects may be relevant, as subsidies for Natura 2000 compared to other subsidies were very low in the 2007-2013 funding period.

The third paper contributes to the understanding of habitat fragmentation vs. habitat loss effects in the context of highway construction. By applying the Schaefer model, we show how wildlife populations will (theoretically) change with respect to habitat loss and habitat fragmentation effects. By drawing from a wide array of ecological literature, we try to clarify the mechanism of how highway construction affects wildlife populations in relation to the spatial distance from the highway. We also link our findings to current discussions on environmental impact assessments.

1.3 Summary of empirical findings

1.3.1 Agri-environmental programs on wildlife

In our first paper, we study the impact of the Austrian agri-environmental program ÖPUL on several wildlife species, particularly roe deer, red deer, wild boar, and brown hare. All of these species require different habitat conditions and have different population histories, therefore, it is not surprising that our findings are diverse.

The effect of ÖPUL funding is positive for *roe deer* in districts characterized by extensive farming. This class represents 48% of all Austrian districts. For *red deer*, we find a positive effect of ÖPUL in 51% of all districts. In contrast to roe deer, red deer seems to benefit from ÖPUL measures particularly in intensively used agricultural areas. In contrast, the effect of ÖPUL on *wild boar* is negative, and it is only significant in intensively used agricultural areas (20% of all districts). Finally, *brown hare* is also negatively affected by ÖPUL measures. The effect is significant in 78% of all districts characterized by less intensive agriculture.

As we have explained, the impacts of ÖPUL funding are complex, as the combination of economic and ecological systems must be considered. This is also reflected in our results, where each effect is different depending on the species, habitat conditions, and agricultural characteristics. They also suggest that for some species, the effect is very small and could only be measured in specific areas, while for other species the effect is larger and measurable across a wide range of districts.

1.3.2 Natura 2000 and land rental prices

The empirical contribution of our second paper is the estimated elasticity of Natura 2000 designation on land rental prices. Given our propensity score approach, our estimation may present a lower bound to the effect; it is possible that unobserved characteristics may still bias our results. The main finding of our second paper is that the impact of Natura 2000 designation on agricultural land rental prices is negative. We confirm a negative elasticity for three types of land, average land (-2.546), grassland (-1.652), and arable land (-2.018). This finding is important, as it indicates that on average, current funding levels do not fully compensate production impairments that are caused by the designation.

1.3.3 Wildlife and highways

While habitat fragmentation has been called one of the most important threats to global biodiversity (Noss, 1991), other authors have suggested that fragmentation effects on ecology could be positive (Fahrig, 2017). We confirm that the effect of habitat destruction, as measured by within-district highway density, on wildlife populations is negative for two out of three species (roe deer *Capreolus capreolus* and wild boar *Sus scrofa*). Using the density of highways in neighboring districts as an indicator for habitat fragmentation, we also find a positive effect of the neighbor highway density on harvest densities of roe deer and wild boar. As in the own-district case, red deer was not affected.

As Austria's accession in 1995 to the EU required Environmental Impact Assessments (EIAs) to be carried out when constructing new highways, we also control for changes in the effects of highways on wildlife before and after 1995. Our results suggest that EIAs have not changed habitat conditions for wildlife with respect to highways for wild boar, and for roe deer and red deer harvest densities have become even lower according to the within-district effect.

Our analysis of wildlife EIAs in Austria reflects on the quality of EIAs, and the methods being used. We find that most wildlife EIAs for highways were carried out qualitatively, with a focus on interviewing close-by hunter organizations. Compensation measures were mainly recommended based on untested assumptions and without any numerical modeling or simulations.

1.4 Methods overview

The methods applied in this dissertation are diverse, but all papers use some form of econometrics. In particular, we use latent class analysis (paper 1), the generalized propensity score matching method by Imbens and Hirano (2004) (paper 2), and dynamic panel data estimation by Arellano and Bond (1991) in paper 3. In the following sections we give a brief overview of the methods applied.

1.4.1 Latent class analysis

Latent class analysis is a special case of the more general class of mixed (or random parameters) models. It is motivated by the assumption that an observed population distribution may be a mixture of several underlying distributions (Greene, 2011, p. 589f). It is particularly suitable when it can be assumed that the population of interest is diverse, but the causes of this diversity are difficult to measure. A common justification of this is preference heterogeneity among different consumers, or unobserved heterogeneity in the characteristics of producers that may lead to different outcomes.

In general, the latent class model is defined as a mixture of two models, (1) an outcome model and (2) a class model that describes the probability of belonging to a specific class. In the two-class case, assuming we observe a mixture of two normal distributions, the contribution of an individual i to the likelihood is

$$f(y_i|class_i = 1) = N[\mu_1, \sigma_1^2] = \frac{exp\left[-\frac{1}{2}(y_i - \mu_1)^2/\sigma_1^2\right]}{\sigma_1\sqrt{2\pi}} \tag{1.1}$$

in class 1 and and

$$f(y_i|class_i = 2) = N[\mu_2, \sigma_2^2] = \frac{exp\left[-\frac{1}{2}(y_i - \mu_2)^2/\sigma_2^2\right]}{\sigma_2\sqrt{2\pi}} \tag{1.2}$$

in class 2 (see (Greene, 2011) for an explanation of the components). The probability of belonging to class 1 and 2 could then be described as λ and $(1-\lambda)$ respectively. The total contribution of individual i to the likelihood is then

$$f(y_i) = f(y_i|class_i = 1)\lambda + f(y_i|class_i = 2)(1-\lambda) \tag{1.3}$$

and the log-likelihood is

$$lnL = \sum_{i=1}^{N} lnf(y_i) \tag{1.4}$$

which can be estimated by maximum likelihood. The latent class model is very flexible in that it allows different specifications of the class model (e.g. a multinomial logit model). It further allows different specifications of the outcome model, and to place restrictions on the parameters in the outcome model (i.e. restrictions on the values parameters in μ_2). The optimal number of classes is still subject to debate, but the standard procedure to decide on the number of classes is to compare models based on information criteria (e.g. Akaike Information Criterion (AIC) and Bayesian Information Criterium (BIC)).

1.4.2 Propensity score with continuous treatments

In response to the lack of methods to study treatment effects where the treatment is continuous and (weakly) ignorable, but where overlap issues exist, Imbens and Hirano (2004) developed a method of continuous propensity score matching. Their method is based on the assumption of weak unconfoundedness of treatment, which is defined as $Y(t) \perp T|X \; for\, all\, t \in T$, where $Y(t)$ is the outcome variable, T is the treatment, and X are observable characteristics of the individual.

The generalized propensity score (GPS) is defined as the *conditional density of the treatment given the covariates*: $r(T|X) = f_{T|X}(t|x)$. The GPS has a balancing property in that within the same strata of X, the treatment variable does not depend on X (Imbens & Hirano, 2004, p. 2). A critical issue in the application of the GPS method is the choice of the propensity score model. Our application is interesting because the treatment is (1) continuous and (2) bounded by the interval $[0, 1)$. Using the zero-inflated Beta (ZIB) model (Ospina & Ferrari, 2010, 2012a, 2012b) for the generalized propensity score fulfills these properties, and allows us to model the two-stage decision of Natura 2000 farming subsidies: first, the decision of whether to subsidize or not, and second the share of subsidized farms in each district. This reflects two different policy levels - i.e. the state level, and the district level. The ZIB model is a mixture of a binary model that models the probability density of a zero vs. non-zero treatment, and a beta distribution model that models the probability density of the treatment on the open interval $(0, 1)$. The Beta density is

$$f\left(N;\mu,\phi\right) = \frac{\Gamma\left(\phi\right)}{\Gamma\left(\mu\phi\right)\Gamma\left(\left(1-\mu\right)\phi\right)}N^{\mu\phi-1}\left(1-N\right)^{(1-\mu)\phi-1},\ N\ \in (0,1) \tag{1.5}$$

and the density of the ZIB model is

$$bi_0\left(N;\alpha,\mu,\phi\right) = \begin{cases} \alpha & if\, N = 0 \\ (1-\alpha)f\left(N;\mu,\phi\right) & if\, N \in (0,1) \end{cases} \tag{1.6}$$

where α is the probability of a zero treatment level. The GPS is predicted for each observation and then observations are matched based on the GPS according to a number of pre-defined classes. After matching, we check the balancing property of the GPS by running t-tests of coverarates between different treatment level groups at given GPS intervals. In the final step, we regress the outcome variable (i.e. land rental price) on the treatment (i.e. Natura 2000 farm share) and the GPS.

1.4.3 Dynamic panel data and wildlife populations

In the third paper, we apply the dynamic panel data estimation method developed by Arellano and Bond (1991). Wildlife harvest may be inherently dynamic in nature, as contemporary reproductive rates depend on previous harvests. Using the first temporary lag of harvest, i.e. the lagged dependent variable, is one method of controlling for previous harvesting. But when using panel data methods to remove unobserved individual time-indifferent effects, endogeneity issues arise. These problems are particularly severe when observations include many individuals and only few time periods.

In the within-model, i.e. the estimation strategy where the individual effect is removed by centering each observation unit around its mean, endogeneity persists because the mean of the within-transformed lagged dependent variable is correlated with the within-transformed error term. Therefore, estimators of the within-model will by biased and inconsistent given the lagged dependent variable. However, as Anderson and Hsiao (1981) find, the first-differenced model can be estimated consistently by IV, using the second lag of the dependent variable (or its first difference) as an instrument for the first lag. As long as the error terms are not serially correlated, this yields consistent, though not necessarily efficient estimators.

A more efficient estimator is the well-known Arellano-Bond estimator (Arellano & Bond, 1991), which also uses first differences to remove the individual-specific effect and uses higher-order lagged dependent variables as instruments to remove the endogeneity problem (see Baltagi, 2005, p. 149f). Arellano and Bond propose to test for second-order autocorrelation in the error terms, as their GMM estimator relies on $E[\Delta v_{it} \Delta v_{i,t-1}] = 0$. From Arellano and Bond (1991, p. 282), the test statistic is defined as

$$m_2 = \frac{\hat{v_{-2}} \hat{v}_*}{\hat{v}^{1/2}} \sim N(0,1) \tag{1.7}$$

where the definition of $\hat{v}$ can be seen in Arellano and Bond (1991, p. 282). In addition, Arellano and Bond suggest the Sargan test of overidentifying restrictions to test if the instruments are valid:

$$m = \Delta\hat{v}'W \left[\sum_{i=1}^{N} W_i'(\Delta(\hat{v}_i)(\hat{v}_i)'W_i \right]^{-1} W'(\Delta\hat{v}) \sim \chi^2_{p-K-1} \tag{1.8}$$

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Chapter 2

Austrian Agri-environmental Programs and Deadweight Losses: A Latent Class Approach

Dieter Koemle and Xiaohua Yu[1]

Abstract

The effects of EU agri-environmental programs (AEPs) on the environment have been mixed. Spending on AEPs has largely been management-based rather than results-based, and they have been described as having ambiguous and unmeasureable goals. In this paper, we study the effects of agri-environmental payments on four wildlife species (roe deer, red deer, wild boar, and brown hare) in Austria. First, we develop a theoretical model to explain the relationship between wildlife and AEPs given rational farmer behavior. We then apply the latent class model to disentangle real ecological impacts of AEPs from deadweight losses. Our results suggest that between 22% (brown hare) and 80% (wild boar) of Austrian districts do not have any significant effects from AE payments. If impacts exist, they can be positive or negative, depending on the species. Based on our results, we recommend a regionalization of agri-environmental payments based on regional agro-ecological characteristics and target species.

Key words: Wildlife; Habitat; Latent Class Analysis; Agri-Environmental Programs; Farmer Behavior

2.1 Introduction

The European Common Agricultural (CAP) emphasizes it's environmental perspectives, as the agri environmental programs (AEP) are an important component of its overall policy design. A large body of literature has shed light on the effectiveness on policy goals of environmental protection (e.g. Uthes & Matzdorf, 2012). The largest

[1] The paper was written by DK. The idea was jointly developed by DK and XY. Data were collected and analyzed by DK. XY provided comments on methodology.

part of commitments related to AEPs are voluntary and will depend on farmers' willingness to participate. Current research suggests diverse motivations for farmers' AEP participation, including farm and program characteristics, as well as individual characteristics of the farmer and the community. However, farmers' participation in an AEP may not necessarily indicate an actual change in the production behavior of the farmer, particularly if the program is designed to induce wide-spread participation by having low requirements. More recently, this common "payments by management" approach to AEPs has been questioned with regard to its effects on improving environmental quality and inducing behavioral changes of farmers (Burton & Schwarz, 2013; Quillérou & Fraser, 2010). As an alternative, researchers and policy makers have experimented with results oriented payment schemes (Burton & Schwarz, 2013; Klimek, Richter gen. Kemmermann, Steinmann, Freese, & Isselstein, 2008; Matzdorf, Kaiser, & Rohner, 2008; Schroeder, Isselstein, Chaplin, & Peel, 2013; Wetzel et al., 2018). However, these schemes suffer from two key obstacles, namely the increased risk for suppliers and the difficulty of developing suitable indicators of success that are cost-effective in monitoring (Burton & Schwarz, 2013; Latacz-Lohmann & Schilizzi, 2007).

As an approach to circumvent the problems caused by results-based payments, designing AEPs based on regional characteristics presents a middle ground between the two above-mentioned approaches. In principle, regionalization of AEPs so far was based mainly on administrative borders, e.g. the national borders in Austria or the federal state level in Germany. However, basing AEPs on administrative borders at this coarse scale may not be suitable for achieving goals of environmental improvement, in particular if the agricultural characteristics within these borders are diverse. Therefore, differentiating AEPs at a lower administrative level, for example based on the agricultural characteristics of a region, could help to increase the effectiveness of AEPs with respect to their policy goals. This paper provides a first step by developing a framework for disentangeling regions where agri-environmental programs may change habitat conditions for certain indicator species (for better or for worse). Within the EU, Austria is among the leading country (together with Luxembourg), investing on average about 135 €/ha agricultural land into AEP as presented in Figure 2.1 (EU Commission, 2017). Austria is an appropriate region to study the specific effects of AEP species. Therefore, we use the Austrian agri-environmental program ÖPUL and its effect on four common wildlife species, roe deer *Capreolus capreolus*, red deer *Cervus elaphus*, wild boar *Sus scrofa*, and brown hare *Lepus europaeus*.

While evaluations of AEPs are mandatory, and many have been published since their first introduction, their focus often lies on evaluating the participation rates and expenditures. Within the AEPs evaluation, the ecological effects though play a minor role. These reports are usually published by the regulating bodies in charge of the rural development programs. As we illustrate in Figure 2.2, the path from the policy design to the actual ecological effect passes several stages: (1) adoption, (2) behavioral change of the farmer, (3) habitat change, and finally (4) population change. If at any point along this chain there is no change induced by the policy, the resulting effect will be zero. Obviously, the outcome could be different for different species. For example, a behavioral change of the farmer may lead to a habitat change for species A, but not for species B. Uthes and Matzdorf (2012) find that the majority of scientific studies on AEPs are ecological in nature. By using experimental or quasi-experimental studies, researchers try to find the impact of a specific agri-environmental measure on a species,

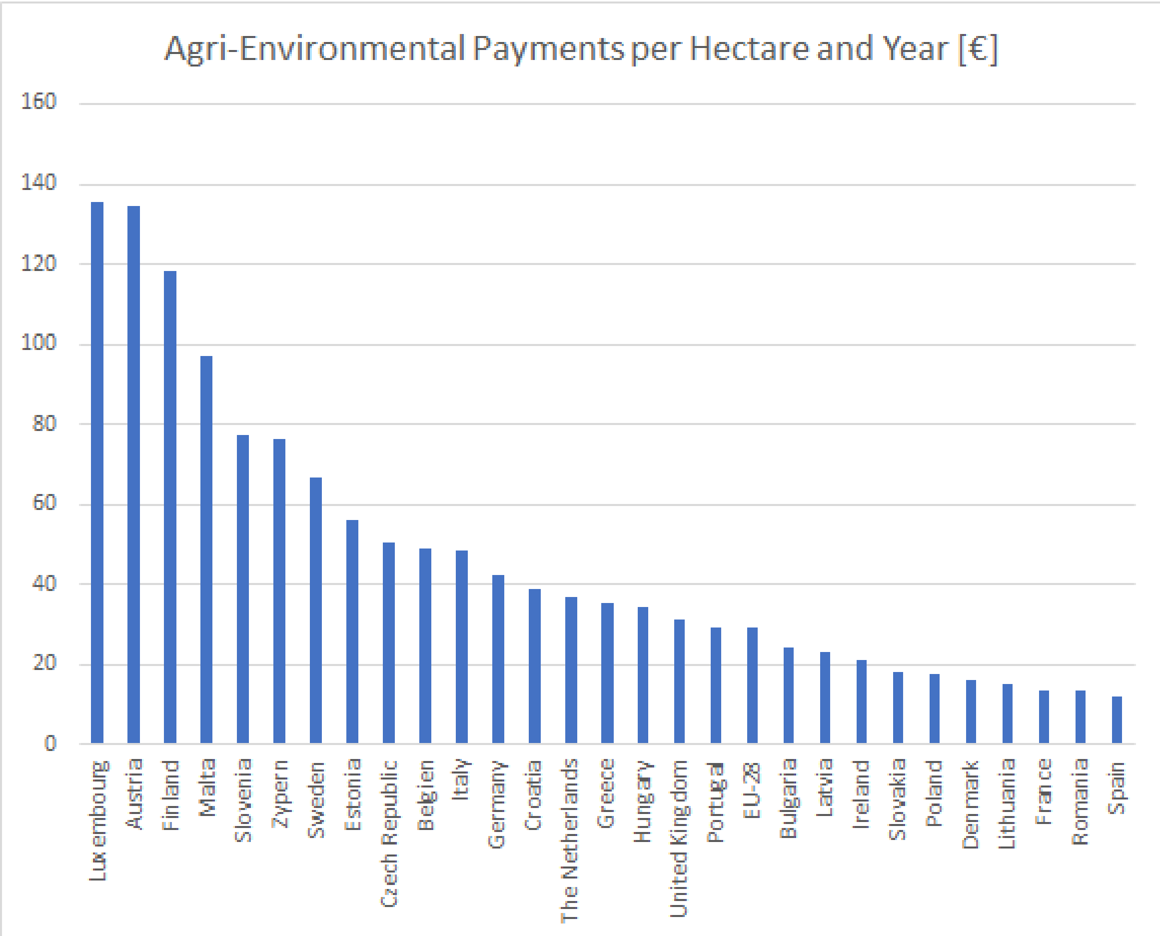

Figure 2.1: Payments for Agri-Environmental Programs 2014-2020 (EUR/ha/year) (Source: own calculations, data from EU Commission 2016)

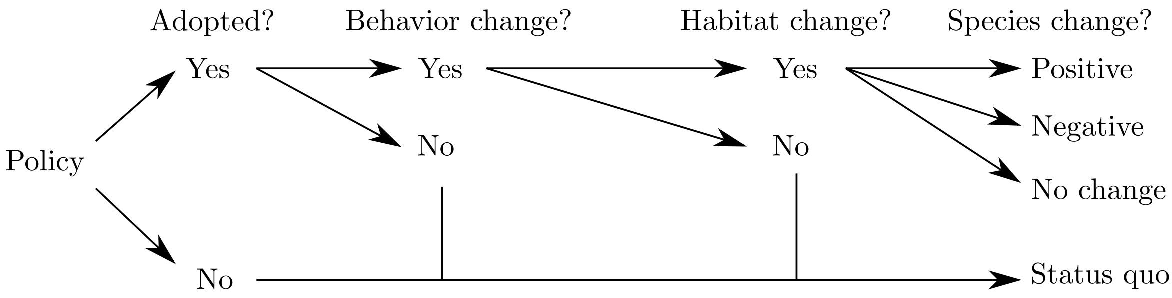

Figure 2.2: Conceptual representation of the path from policy design to the ecological effect

or biodiversity in general. In Figure 2.2, this refers to the right-hand side. A second field of research studies the adoption motivations of farmers, i.e. the left-hand side in Figure 2.2. This section of our conceptual framework also includes research studying the public and private transaction costs from agri-environmental programs. The gap between these two sides, the actual behavioral change of the farmer that may lead to a habitat change, is the research subject of the present paper.

Apart from its conceptual appeal, our paper has practical implications for future policy design. Wildlife species in Europe, from large ungulates to smaller game, have been strongly influenced by human interference. They have been hunted for centuries, and their populations have been influenced by infrastructure (Koemle, Zinngrebe, & Yu, 2018) and the structural change of agriculture, as it could be observed in Eastern Europa after the fall of the iron curtain (Batáry et al., 2017; Donald, Sanderson, Burfield, & van Bommel, 2006, 3-4). Though intensive agricultural practices in the modern era have impoverished seed banks, reduced biodiversity, and caused landscape fragmentation (Gliessman, 2014; Uthes & Matzdorf, 2012), some species in Europe have rapidly expanded in recent decades due to reductions in hunting pressure, the absence of large predators, and the colonization of modified habitats (Hewison et al., 2001). Other species have decreased in population. The three ungulates represent the most important game animals in Austria, while brown hare has been used as an indicator of environmental quality in many studies. Moreover, each of these species has a particular preference profile regarding its habitat. Apart from their ecological functions and the revenue they bring to hunters, these wild game species are also economically relevant due to damage they cause in agriculture and forestry (Bleier, Lehoczki, Újváry, Szemethy, & Csányi, 2012; Reimoser & Gossow, 1996; Reimoser & Putman, 2011; Verheyden, Ballon, Bernard, & Saint-Andrieux, 2006), and due to traffic accidents involving wildlife (Langbein, Putman, & Pokorny, 2010; Putman, 1997; Steiner, Leisch, & Hackländer, 2014). As deer numbers have increased in much of Europe, these issues are likely to gain importance in the future (Milner et al., 2006). It is therefore important to study how AEPs affect game species in the context of a given (agro-)ecosystem.

2.2 Background and Literature Review

In this paper we study the link between agri-environmental programs (AEPs) and wildlife populations. Environmental policy enters agricultural policy through both CAP pillars, the first pillar via Cross Compliance (i.e. constraints on production linked to the first pillar direct payments), Greening payments and through the agri-environmental programs (AEPs) within the rural development programs (second pillar). In the following sections we first outline the literature on participation in AEPs and then we explain how wildlife populations of our indicator species could be affected by them.

2.2.1 Farmer participation in Agri-Environmental Programs

A wide array of studies has conducted research on the participation decisions of farmers in agri-environmental programs, using stated- as well as revealed preference methods. By using a panel data approach based on the random effects logit model, Hynes and Garvey (2009) find that farmers that are already using extensive production systems are most likely to participate in agri-environmental programs. This finding is also supported by Zimmermann and Britz (2016), who study determinants of farmer participation in AEPs using the Heckman two-step model. They base their analysis on the FADN dataset comprising of farm-level data of 22 member states over the period 2000-2009. The authors find that higher shares of grassland is correlated with a higher share in AEP participation, while farmers with more intensive uses such as vegetable production are less likely to participate. Similarly, Defrancesco, Gatto, Runge, and Trestini (2008) find that labor-intensive farming practices, investment-intensive farming systems and a strong dependence on farming income reduce the likelihood of participation in AEPs. Low subsidies are generally seen as a deterrent from participation as well es for the extent of AEP adoption (Giovanopoulou, Nastis, & Papanagiotou, 2011).

Apart from the farm characteristics, also other factors may influence participation decisions, including the farmer's age and education (Defrancesco et al., 2008; Giovanopoulou et al., 2011), as well as relationships with neighbors, the farmers' environmental goals and the vision about the future about the business (Defrancesco et al., 2008). Using a stated preference method, Ma, Swinton, Lupi, and Jolejole-Foreman (2012) study the determinants of participation in a hypothetical AEP using a double-hurdle model on survey response data. They find that farmers will only consider participating if the payment is high enough, and if the farmers believe this payment to be politically feasible. After that, they will only enroll if the utility gain from participation is higher than their willingness to accept.

2.2.2 The Austrian Agri-Environmental Program ÖPUL

The Austrian agri-environmental program (ÖPUL) was introduced prior to Austria's accession to the European Union (Kleijn & Sutherland, 2003). It is co-funded by second-pillar Common Agricultural Policy (CAP) payments that compensate farmers for voluntarily practicing organic or integrated farming, planting catch crops and flowering strips, raising endangered livestock species, or improving water protection. The ÖPUL program of 2007-2014 consisted of 29 measures (Rechnungshof, 2013). While each measure has defined environmental goals, the Austrian Court of Audits (Rechnungshof) has criticized that the goals of ÖPUL are often formulated too broadly, and lack of data would make the evaluation of goal achievements difficult (Rechnungshof, 2013, p. 303).

The largest part of ÖPUL funding (20.6%) was spent on the measure **"environment-**

friendly farming on arable land and pastures" (UBAG), which was adopted by 38% of Austrian farmers (Rechnungshof, 2013). This measure aims at reducing the use of fertilizers and pesticides on arable lands and pastures, and at protecting traditional landscape elements according to Annex F of the ÖPUL Directive (e.g. single trees, hedgerows, small ponds, stone walls, meadow orchards, marshes and dry grasslands, and others). Nitrogen-fertilizer is restricted to 150 kg/ha and livestock limited to 2 livestock units per ha. Farmers with more than 5 ha of arable land are required to keep at least 25% free of grains, maize production, and fallow land kept in good agricultural and ecological condition. No crop is allowed to cover more than 66% of arable land. Farmers must add 2.5m to 12m wide strips of "biodiversity enhancing areas" to their arable land (2%-5% of the total arable land), which has to be seeded with a seed mix before May 1st and can be chaffed earliest in August. Spraying equipment needs to be checked professionally on a regular basis. The funding per hectare and year varies between 50 € and 100 €. UBAG participation is scattered all over Austria, with an emphasis on the northern and north-eastern districts.

The second-largest amount of funding (17.9%) went to **organic farming**, which was adopted by about 12% of farmers. The goal of organic farming is to increase animal and plant biodiversity by applying wide and diverse crop rotations, to reduce the use of synthetic pesticides, and to improve livestock conditions. Organic farming has similar restrictions as UBAG, but is also subject to the directive EEC No. 834/2007 on organic farming, including stricter regulations on the use of synthetic fertilizers and chemical plant protection within the production process, and a prohibition of ionizing radiation of seeds and the use of genetically modified organisms. Grassland can be reduced at most 5% relative to the first year of participation. Similar to UBAG, farmers are required to keep flowering margins on their arable land, with the same restrictions on chopping and ploughing. Livestock farmers need to provide their roughage consuming cattle with hay in addition to silage. In addition, participating in organic farming requires farmers take at least 15 hours of training courses including excursions. Funding on arable land varies between 110 €/ha (feed crops) and 600 €/ha (vegetables) per year; funding on pastures varies between 110 €/ha per year and 240 €/ha per year; funding for vineyards and fruit plantations is 750 €/ha per year; and funding for products grown in greenhouses can go up to 4200 €/ha/year. Organic farming is largely concentrated in the central, alpine dominated districts of Austria.

The third largest amount of ÖPUL funding (12.5%) was spent for planting **catch-crops on arable land**, which was adopted by 29% of farmers. Farmers could choose from seven different catch-crop options differing in plant variety, seeding and cutting time, planting and restrictions on the use of herbicides. Subsidies range from 130 to 190 €/ha/year. Participation in catch cropping was particularly high in districts with a large share of arable land.

Farmers who did not participate in organic farming (which is a top-up to the UBAG measure), could also choose more specific measures to improve their environmental performance and acquire additional subsidies. These include e.g. the (1) "restricted use of fertilizers and pesticides on arable land" (115 - 165 €/ha/year), (2) "restricted use of fertilizers and pesticides on pastures and feed crops" (50 €/ha/year), and (3) "restricted use fungicides on grain crops" (25 €/ha/year). All three measures restrict the use of fertilizers and pesticides to those permitted according to EEC No. 834/2007 (organic farming), but they may be applied individually.

In 2010, ÖPUL has been adopted by 116.122 (67%) of Austrian farmers (Rech-

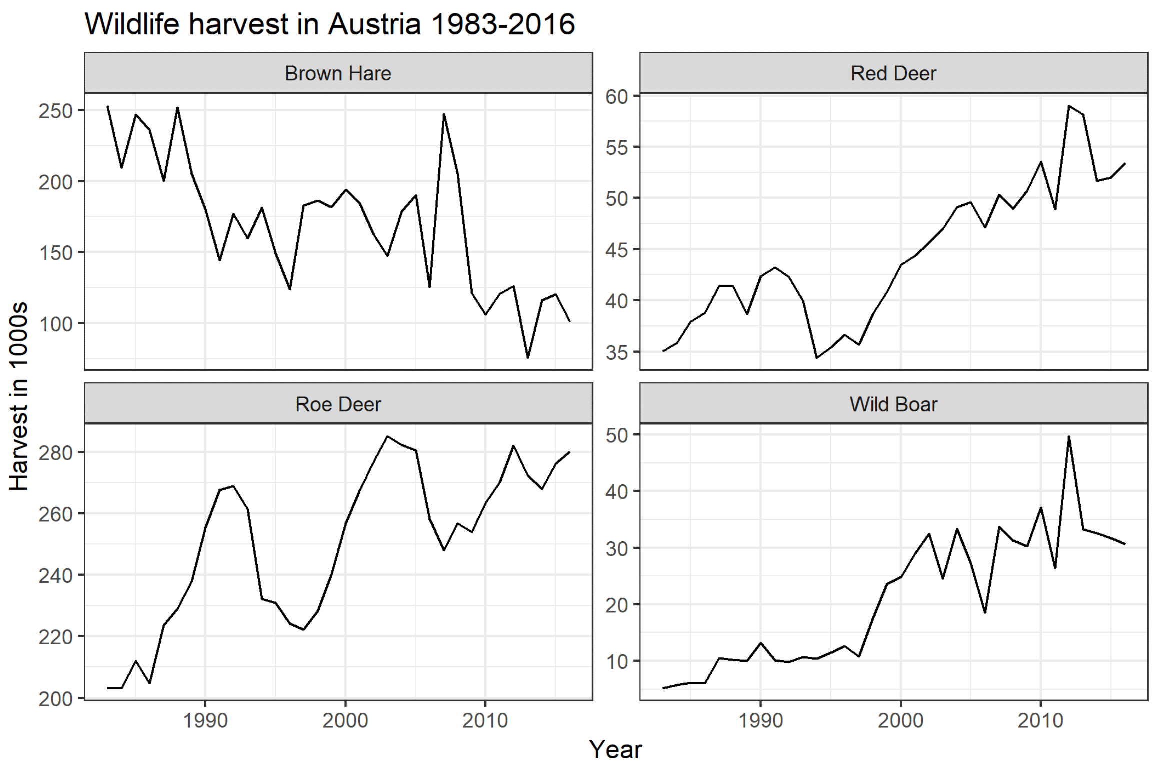

Figure 2.3: 1983-2016 total annual harvest of red deer, roe deer, wild boar, and brown hare in Austria (Source: Statistik Austria)

nungshof, 2013). However, it remains unclear whether this wide acceptance has been mainly driven by the low requirements and loose production constraints (i.e. deadweight loss in terms of conservation improvement goals), or whether farmers were also motivated to change their production behavior.

2.2.3 Wildlife Management in Austria

Wildlife management in Austria is mainly conducted by the hunting associations, and characterized by closely regulated hunting, strongly circumscribed administrative requirements and constraints as well as traditional practices (Putman, Apollonio, & Andersen, 2011). Hunters are required to renew their hunting licenses every year.

Figure 2.3 shows the total annual harvests of roe deer, red deer, wild boar, and brown hare in Austria. In numbers, roe deer harvest is highest, totaling 280.000 in 2016. The harvest numbers of red deer and wild boar were 53.000 and 30.000 respectively in 2016. Brown hare harvest was 101.000 in the same year. Figure 2.4 presents the average harvest density distributions over the study period (2005-2014). Red deer is concentrated in the mountainous West and center of Austria, where agriculture is dominated by extensive livestock farming, pastures, and forestry. Roe deer and brown hare reach their highest densities in the flat to hilly North and East, also reach sizable densities in mountainous areas. Finally, wild boar densities are the highest along the Eastern border.

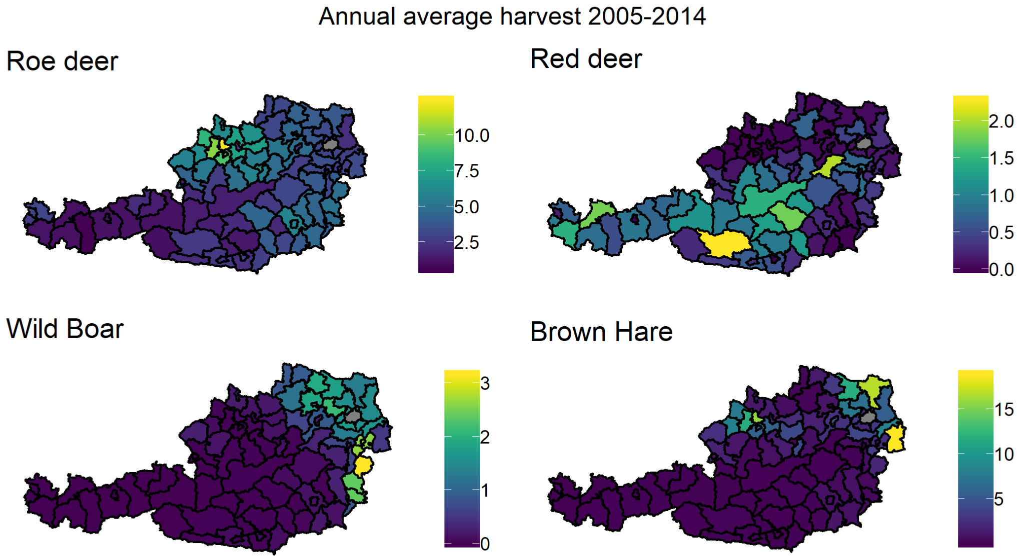

Figure 2.4: Average annual harvest density 2005-2014 in Austrian districts (Garnier, 2018; Wickham, 2016)

2.2.4 Agriculture and Wildlife

As a side effect, few studies have investigated the effect of agri-environment schemes on mammals. Agri-environmental policies inevitably change food sources and habitats for animals, though different measures may have different effects on different species.

For instance, UBAG could provide habitats of better quality in agricultural areas through the introduction or preservation of landscape elements, but it also limits fertilizer application of farmers which could reduce food sources for wildlife species. Similarly, organic farming practices involving less synthetic fertilizers and pesticides often produce lower agricultural outputs, but require more labor inputs, so that they could both reduce the feed sources from agriculture, but also increase human disturbance due to mechanical weeding. A reduction or abolition of synthetic fertilizers and pesticides may also affect wild ungulates physically. Research in several western countries links mammal deaths to the application of insecticides, rodenticides, molluscicides, herbicides, and fungicides (Berny, 2007). Despite limited information on the toxicity of herbicides for mammals, herbicides have been suspected to pose a direct hazard to herbivores; fungicides have been found to be of limited importance.

Red deer is considered as an intermediate feeder, choosing a mixed diet of grass, sedges, and concentrate food (Gebert & Verheyden-Tixier, 2001; Hofmann, 1989). Given that in some countries, red deer use agricultural land for feeding, particularly female red deer may prefer well-fertilized agricultural meadows over unfertilized meadows and forage available in forest habitats (Lande, Loe, Skjærli, Meisingset, & Mysterud, 2013; Zweifel-Schielly, Leuenberger, Kreuzer, & Suter, 2012). However, whether red deer is sensitive to agricultural practices will depend on how it utilizes agricultural land. Feeding, as it is practiced in Austria to reduce forest and agricultural damages (Schmidt, 2014) may reduce red deer's dependence on agricultural lands, and therefore diminish any impacts caused by agri-environmental policy.

Similar to red deer, roe deer is essentially a woodland species (Hewison et al., 2001). Roe deer are concentrate selectors (Hofmann, 1989) and may have successfully colonized agricultural landscapes to gain access to highly energetic and digestive resources (Ferron et al., 2011). Further, landscape elements such as hedgerows between fields and meadows can act as substitutes for woodlands, which provide roe deer with cover from weather and predators, including humans (Morellet et al., 2011). We hypothesize that measures such UBAG, which are also aimed at improving landscape attractiveness through hedgerows and the availability of grasslands, could influence roe deer stock levels positively. On the other hand, measures that reduce food availability, may have a negative influence on roe deer.

Wild boar is considered an omnivorous, opportunistic species, with a diet consisting of 90-99% of plant matter (Amici, Serrani, Rossi, & Primi, 2011; Cuevas, Ojeda, Dacar, & Jaksic, 2012; Herrero, Irizar, Laskurain, García-Serrano, & García-González, 2005). While energy-rich plant matter such as acorns, or beech and oak mast are the preferred food of wild boar, agricultural crops seem to be an important food source throughout Western Europe (Schley & Roper, 2003). The availability of agricultural crops has been found to have a significant effect on body condition of piglets (Merta, Mocala, Pomykacz, & Frackowiak, 2014). Similar to roe deer, wild boar could be sensitive to reductions in fertilizer application, leading to better body conditions of piglets and increasing their change of survival. Finally, wild boars have been found to move along linear landscape elements between fields. Providing these elements under UBAG could improve conditions for wild boar and lead to increased stocks (Thurfjell et al., 2009).

Austrian populations of brown hare have been declining over the past decades (Bauer, 2001). Bauer points out that declines have been largely caused by transformation and uniformization of agricultural land and the widespread application of agrochemicals.

Given different physiological and behavioral characteristics of red deer, roe deer, wild boar, and brown hare, means that they also occupy different ecological niches and have different habitat requirements. The (agro-)ecosystem in which a species can thrive may be related to food and shelter availability, previous hunting pressure, (human) disturbance through noise, emissions, and other factors. We therefore separately estimate the unintended impacts of the Austrian agri-environmental program on these species.

2.3 Theoretical Framework

Our theoretical model combines two components, (1) the farmers' behavior and (2) the ecological response.

2.3.1 Farmer decisions

We assume a profit maximizing farmer with optimal resource allocation decisions. The farmer will participate in an agri-environmental program if the marginal benefits of participating (i.e. the subsidy) outweigh the additional costs (or forgone profits) incurred from the program.

$$\pi_{it} = pq\left(L\right) + s\left(L\right)L - wx\left(L\right) \tag{2.1}$$

Where q is a vector of outputs, p are corresponding output prices, x is a vector of inputs and w are input prices. We assume that input and output prices are exogenous.

Assume, for simplicity, a single agri-environmental subsidy s that is allocated to a land area $L \leq \bar{L}$, where $\bar{L}$ is the total area of land available to the farmer. Output q is a function of L, and generally $\frac{dq}{dL} \leq 0$ because agri-environmental programs generally restrict farmers' production decisions. Nevertheless, as some AEP measures include farmer training and education, $\frac{dq}{dL}$ could also be positive if the farmer has not been fully efficient given his natural capacities due to lack of knowledge. However, in most cases it is likely that additional training may help to mitigate productivity losses, but not fully compensate for them. The choice of inputs is restricted by L, but the sign of $\frac{dx}{dL}$ will depend on the specific input. For example, it could be negative for synthetic fertilizer, but positive for labor. The first order condition with respect to the agri-environmental program is then

$$\frac{\partial \pi_{it}}{\partial L} = p\frac{dq}{dL} + s(L) + \frac{ds}{dL}L - w\frac{dx}{dL} = 0 \tag{2.2}$$

The subsidy is endogenous with land, because not only will the size of the subsidy influence the amount of land in the program, but also the program designers will try to anticipate how many farmers will pick up the program due to budget limitations. We can solve this result for L to see how the land under AEPs depends on the subsidy and input and output prices:

$$L = \left(w\frac{dx}{dL} - \left(p\frac{dq}{dL} + s(L)\right)\right)\frac{dL}{ds} \tag{2.3}$$

We can assume that $\frac{dL}{ds} > 0$, as it is plausible that a higher subsidy will lead to a higher uptake of AEP measures and vice versa. Intuitively, uptake will be positive if and only if

$$w\frac{dx}{dL} < \left(p\frac{dq}{dL} + s(L)\right) \tag{2.4}$$

More clearly, the magnitude of L therefore critically depends on the relationship between the subsidy, and the marginal revenue and marginal cost:

$$s(L) \geq -p\frac{dq}{dL} - w\frac{dx}{dL} \tag{2.5}$$

Case 1: $\frac{dx}{dL} > 0$: the subsidy needs to be larger than the losses in revenue, as it must compensate for the increased cost.

Case 2: $\frac{dx}{dL} < 0$: because the cost decrease compensates for the revenue loss, the subsidy can be lower than in the case above and still produce a positive AEP adoption.

Finally, if prices were not fixed, we see that AEP participation would increase with increasing input prices and decrease with increasing output prices.

In principle, the model shows the intuitive result that (1) high subsidy, (2) low losses in marginal revenue, and (3) decreases or low increases in marginal cost will lead to a higher adoption of AEPs. Given that the per hectare subsidy is the same, farmers in marginal areas are more likely to face lower losses in marginal revenue and may possibly even face positive or zero changes in marginal costs. The arguments outlined above can be summarized in three farmer categories:

1. Marginal farmers: Some farmers may be able to add the full subsidy to their annual profit without any change in their production behavior; i.e. input constraints are not binding. In this case, $\frac{dq}{dL} = \frac{dx}{dL} = 0$. The AEP will have no effect on the habitat characteristics of the farmland. This is a typical case of adverse selection (Quillérou and Fraser, 2010).

2. Medium productivity farmers: These farmers change their production behavior as a result of the AEP participation. The additional profit from the subsidy is still large enough for them to participate, i.e. $s\left(L\right) > -p\frac{dq}{dL} - w\frac{dx}{dL}$.

3. High productivity farmers: High productivity farmland may be affected severely by participation in AEPs. Keeping with the example of fertilizer, a limit in synthetic fertilization may severely reduce output, but also increase the cost of additional supportive measures such as tillage or mechanical weeding. Therefore, the per-hectare subsidy will be a smaller share of per hectare revenue than in the two former categories. Farmers in this category are unlikely to participate in an AEP. The cutoff will be where $s\left(L\right) = -p\frac{dq}{dL} - w\frac{dx}{dL}$.

From the three categories above, only the second category will change their behavior and therefore habitat conditions for wildlife. Nevertheless, category 1 may also provide good conditions for wildlife populations. The ÖPUL program offers an additional complication, as farmers can only subscribe their entire cultivated farmland to a measure. This means that even within a single farm, farmland of the three different categories may exist. A farmer will then participate if the benefits from the "winner" farmland outweigh the additional costs from the farmland that loses profitability from the policy. We illustrate the uptake of AEPs in Figure 2.5.

2.3.2 Wildlife response agri-environmental programs

Wildlife population dynamics can be modeled through the logistic growth model

$$\frac{dN}{dt} = N_t r \left(1 - \frac{N_t}{K}\right) - H_t \tag{2.6}$$

Where H_t is annual harvest, N_t is population size, r is the intrinsic growth rate and K is the carrying capacity. It is reasonable to assume that the parameters r and K will change with respect to the available feed and cover conditions through land under AEPs. In equilibrium

$$N_t r\left(L\right) \left(1 - \frac{N_t}{K\left(L\right)}\right) = H_t = qN_tE_t \tag{2.7}$$

Where the right-hand side is the well-known Schaefer model (Conrad & Clark, 1987; Schaefer, 1957) that describes harvest as a function of stock N, effort E, and the catchability coefficient q. We can solve 2.7 for stock to yield

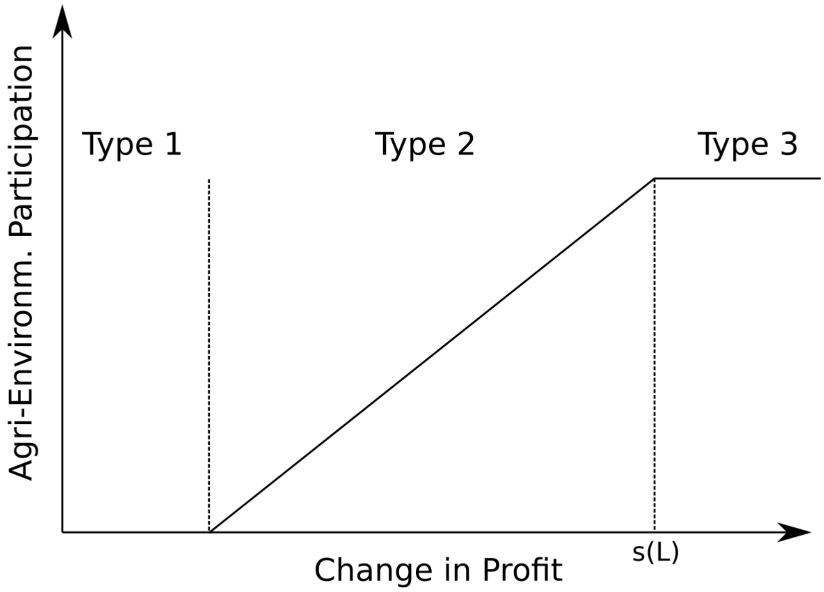

Figure 2.5: Participation in agri-environmental programs as a function of profit change for different types of farmers

$$N_t = K\left(L\right)\left(1 - \frac{qE_t}{r(L)}\right) \tag{2.8}$$

The sign of $\frac{dK}{dL}$ may be positive or negative, depending on the species and the specific program. Therefore if $\frac{dK}{dL} > 0$, stock and, according to the Schaefer model, annual harvest will increase.

Case 1: $\frac{dr}{dL} > 0$ and $\frac{dK}{dL} > 0$: Harvest will increase as both the intrinsic growth rate and carrying capacity increase.

Case 2: $\frac{dr}{dL} < 0$ and $\frac{dK}{dL} > 0$: The magnitude of $\left(1 - \frac{qE_t}{r(L)}\right)$ will decrease. Therefore, the population effect depends on the relationship between the marginal changes of carrying capacity and intrinsic growth rate.

Case 3: $\frac{dr}{dL} > 0$ and $\frac{dK}{dL} < 0$: Similarly to case 2, harvest changes depend on the relationship of marginal changes of the population parameters.

Case 4: $\frac{dr}{dL} < 0$ and $\frac{dK}{dL} < 0$: Annual harvest will decrease.

2.4 Econometric model

Our theoretical model suggests that there are three types of farmers. However, in practice, the separation of farmer types may be endogenous, as not all of the farmers' relevant characteristics are observed as is usually the case in production based models (Ackerberg, Lanier Benkard, Berry, & Pakes, 2007). This is because the explanatory variables in the farm's production function are generally chosen by the farmer, who may have knowledge of the farm's unobserved characteristics (e.g. soil quality, landscape and plot structure, and other characteristics). This leads to endogeneity problems when estimating the harvest equation, because the farms' unobserved production characteristics introduce a correlation between the explanatory variables and the error term. Therefore, an OLS regression of harvest on AEP subsidy will produce biased parameter estimates of the marginal effect of the AEP on wildlife harvest.

An econometric representation of the Schaefer model is

$$H_{it} = qN_{it}E_{it}e^{\varepsilon_{it}} \tag{2.9}$$

Or, taking logs,

$$lnH_{it} = lnq + lnN_{it} + lnE_{it} + \varepsilon_i$$

(2.9) can be regarded as a production function, where stock N is a function of agri-environmental policy uptake L and other farm- and landscape-specific characteristics Z. Hunting effort E could be proxied by the annual number of hunting licenses issued in district i. The endogeneity problem from the farm profit function is carried through to (2.9). Substituting N by the result from (2.8) yields

$$lnH_{it} = lnq + lnK\left(L_{it},\ Z_{it}\right) + ln\left(1 - \frac{qE_{it}}{r(L_{it},\ Z_{it})}\right) + lnE_{it} + \varepsilon_{it} \tag{2.10}$$

Or, more generally,

$$lnH_{it} = lnq + f\left(L_{it},\ Z_{it}, E_{it}\right) + lnE_{it} + \varepsilon_{it} \tag{2.11}$$

Where $f\left(L_{it},\ Z_{it}, E_{it}\right) = lnK\left(L_{it},\ Z_{it}\right) + ln\left(1 - \frac{qE_{it}}{r(L_{it},\ Z_{it})}\right)$. Because farmer decisions on L may depend on unobserved characteristics, $f\left(L_{it},\ Z_{it}, E_{it}\right)$ is likely correlated with ε_{it}. Assume that ε_{it} can be decomposed into a class-specific effect μ_c and an i.i.d. error term ω_{it}, where μ_c controls for the farm-specific ratio between marginal value product and marginal cost

$$lnH_{cit} = lnq + f\left(L_{it}(\mu_c),\ Z_{it}, E_{it}\right) + lnE_{it} + \omega_{it} \tag{2.12}$$

Therefore, if one can successfully account for μ_c, that is, account for unobserved differences between farms that influence AEP participation decisions, it is possible to identify the model outlined in (2.9). We assume that the farmers belong to different latent classes, as outlined in the previous section. Latent class analysis allows us to separate observations into different types (classes).

Econometrically, this model can be estimated by maximum likelihood. In the context of agri-environmental programs, the latent class model has often been applied to elicit farmer preferences for the characteristics of environmental programs, in particular when analyzing stated preference studies such as choice experiments (Glenk & Colombo, 2011; Ruto, Garrod, & Scarpa, 2007; Scarpa & Thiene, 2005) or in farm productivity analysis (Alvarez & del Corral, 2010). The latent class model combines a class membership model with a (mixed) linear model. The class membership model

$P(c_i = g)$ can be defined as a multinomial logit model with separate explanatory variables. The model conditional on class g becomes (Proust-Lima, Philipps, Diakite, & Liquet, 2018; Proust-Lima, Philipps, & Liquet, 2017)

$$H_{it}|_{c_i=g} = x_{1i}\beta_g + x_{2i}\gamma + \varepsilon_{ij} \tag{2.13}$$

where γ describes effects that are identical over all classes and β_g describes effects that are class-specific including a class-specific intercept. For a normally distributed error term, the contribution of an individual at time t to the likelihood is

$$f\left(H_{it}|class = g, X\right) = N\left[\mu_{ig}, \sigma_g^2\right] = \frac{exp\left(-\frac{1}{2}\left(H_i - x_{1i}\beta_g - x_{2i}\gamma\right)^2/\sigma_g\right)}{\sigma_g\sqrt{2\pi}} \tag{2.14}$$

In the panel data case, the log likelihood is

$$lnL = \sum_{i=1}^{n} \ln\left[\sum_{g=1}^{G} p_{ig}\left(\delta, z_i\right)\prod_{t=1}^{T} f\left(H_{it}|class = g,\ x_{it}, \beta_g, \gamma\right)\right] \tag{2.15}$$

Estimation of the model (2.15) was done in the R (R Core Team, 2014) package `lcmm` (Proust-Lima et al., 2018; Proust-Lima et al., 2017). `lcmm` provides a flexible framework to estimate parametric models with fixed and random parameters including latent classes by maximum likelihood, using a modified Marquardt algorithm with strict convergence criteria.

2.5 Data

Austria is separated into nine provinces and 95 individual districts. All estimations were conducted with district level data. Several data sources were accessed and combined to conduct this analysis. A summary of the data is provided in Table 1.

- The statistical bureau of Austria provides annual harvest numbers at the level of the 95 political districts in Austria. Total harvest numbers were transformed into densities (harvest/km^2).
- The numbers of hunting licenses at district level were gathered from the provincial hunting associations and provincial administrations.
- Annual data on ÖPUL payments were provided by the Federal Institute of Agricultural Economics (AWI) at the district level, and were normalized into payment per km^2 of agricultural land.
- The AWI also provides NUTS3 level average farm book-keeping data including annual land use (grains, grasslands, etc), yields, and farmgate prices.
- Data on monthly temperatures and precipitation from all measuring stations for the years 1994-2015 were collected from the Austrian Agency for Meteorology and Geodynamics (ZAMG). We used inverse distance weighting (Pebesma, 2004) to calculate annual average minimum temperatures and precipitation for each district and year in R.

- Data on tree density in 2012 were acquired from Copernicus (www.copernicus.eu) in a 100m by 100m resolution and aggregated to district average levels in QGIS (QGIS Development Team, 2015).

Table 2.1: Summary of the data used to analyze the impact of agri-environmental policy measures on ungulate populations in Austria

Statistic	Unit	N	Mean	St. Dev.	Min	Max
Roe deer harvest	Harvest/km^2	1,034	3.621	2.17	0.553	12.888
Red deer harvest	Harvest/km^2	1,034	0.441	0.545	0.0	4.559
Wild boar harvest	Harvest/km^2	1,034	0.497	0.84	0.0	4.804
Brown hare harvest	Harvest/km^2	1,034	2.291	4.221	0.0	34.878
Farm premium	€/km^2	940	19,988	9,014	1,176	34,532
ÖPUL payment	€/km^2	1,034	19,420	6,063	6,269	44,135
Avg. min. temp.	°C	1,034	-2.641	1.749	-8.867	1.54
Precipitation	cm	1,034	82.84	19.244	32.163	153.73
Altitude	1000 m	1,034	0.729	0.485	0.122	2.098
Hunting licenses	#/km^2	876	1.983	2.134	0.0	12.454
Grassland	ha/farm	1,034	23.113	19.661	0.0	129.98
Grain crops	ha/farm	1,034	8.356	8.363	0.0	42.85
Sugar beets	ha/farm	1,034	0.512	1.02	0.0	5.03
Maize	ha/farm	1,034	2.539	1.951	0.0	8.77
Avg. tree density	Index	1,034	45.59	15.333	5.152	78.149
Unproductive land	%	1,034	0.1	0.16	0.004	0.829
Average farm size	ha	1,034	29.741	11.718	0.0	107.17

Source: own calculations

2.6 Results

We use latent class analysis to disentangle the different effects of agri-environmental payments on wildlife species. Our indicator species roe deer, red deer, wild boar, and brown hare, show different responses to ÖPUL payments, as can be seen in figure 2.7.

2.6.1 General estimation results

Populations of wildlife species are affected by a multitude of factors, and we first discuss the influence common to all species before describing our class-specific findings in more detail. As is common, we used the information criteria AIC and BIC to identify the optimal number of classes in our estimation (Greene, 2011). Table 2.3 presents model statistics and class membership probabilities of four models estimated with three latent classes. According to both AIC and BIC, the three-class specification is superior in all four models. Unfortunately, higher-class specifications did not produce any estimation results. We present the parameters of the latent class estimation in Table 2.2.

For all species except brown hare, the average minimum temperature is negatively related to harvest density. The average precipitation is positively related to all species except wild boar, while all species except red deer seem to prefer habitat of lower altitudes. Larger average amounts of agricultural grassland are related positively to roe deer and brown hare harvest densities, but negatively to red deer and wild boar. Grain crops are positively related to all species except for red deer. Sugar beet is positively related to all species except roe deer. Surprisingly, maize cultivation is negatively related to all species except roe deer. This stands in contrast with research suggesting that maize is an important feed source of wild boar (Herrero, García-Serrano, Couto, Ortuño, & García-González, 2006). Interestingly, tree density shows a negative correlation with all species except for red deer, whereas unproductive land seems to benefit the populations of roe deer and red deer, while wild boar and brown hare show negative correlations with this factor. Average farm size, as an indicator of the agricultural structure in a district, has a negative relationship with roe deer and brown hare, but a positive relationship with red deer and wild boar. Finally, we added a dummy to account for the effect that the new funding period after 2006 has also brought some changes to the agri-environmental measures. It turns out that, given all other factors, harvest densities have been lower after 2006 for roe deer and brown hare, but higher for red deer and wild boar.

2.6.2 Class specific results

The classification of Austrian districts into separate groups with regard to the effect of agri-environmental subsidies and farm premiums produced some interesting patterns, as is shown in Figure 2.6. For all species, we only find evidence of significant impacts of ÖPUL payments in Class 1, accounting for 20% (wild boar) to 78% (brown hare) of all districts (Table 2.3). The signs and magnitudes of the effect differ by species.

In particular, we find a positive relationship between ÖPUL payments and the harvest densities of roe deer and red deer, and a negative relationship for wild boar and brown hare. The direction of this relationship is expressed more clearly in the scatterplots presented in Figure 2.7.

Roe deer

The descriptive statistics separated by latent classes of roe deer are shown in Table 2.4. The highest harvest density of roe deer is found in class 3 (7.9 animals/km^2). Class 3 has the smallest farm sizes (23.8 ha), lowest average altitude (475 m), lowest grassland (16.4 ha/farm) and unproductive land (0.02 ha/farm) and highest maize (3.5 ha/farm), and farm premium. Interestingly, class 3 has the lowest density of hunting licenses (1.7/km^2). Class 2 has the second highest harvest density of roe deer (4 animals/km^2), therefore half as many as class 3. It is higher in average altitude (641 m) and in grassland (20 ha/farm) and highest average farm size (31 ha) in grain crops (10.7 ha/farm), sugar beet (0.8 ha/farm) and tree density (45%). Class 1, finally, has the lowest harvest density (2.3 animals/km^2), and also the lowest farm premium compared to the other two classes. Given the high average altitude (803 m) and grassland (27 ha/farm), and low sugar beet, grain crops, and maize, this reflects that class 3 districts represent regions dominated by extensive farming methods. Therefore, roe deer densities may benefit from smaller-structured farms with higher amounts of grain and maize within low altitude landscapes.

The effect of the farm premium is positive all three classes, but the magnitude differs. It is lowest in class 1, second-highest in class 2 and highest in class 3. This result is intuitive, as Austria has adopted the historical model of pillar 1 direct payments,

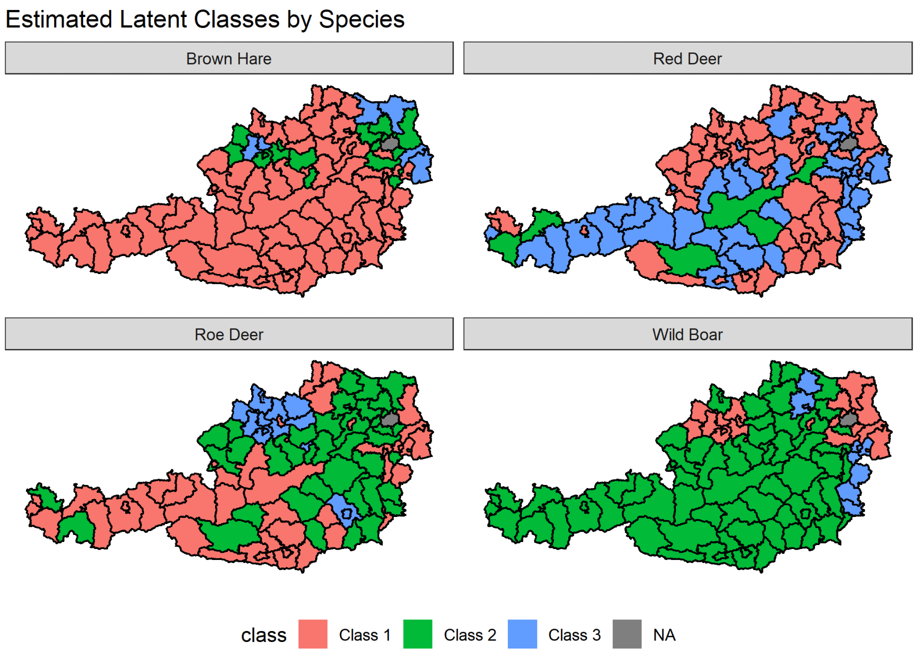

Figure 2.6: Latent classes of districts regarding their effect of agri-environmental subsidies on wildlife species. Note that the classes cannot be compared across species.

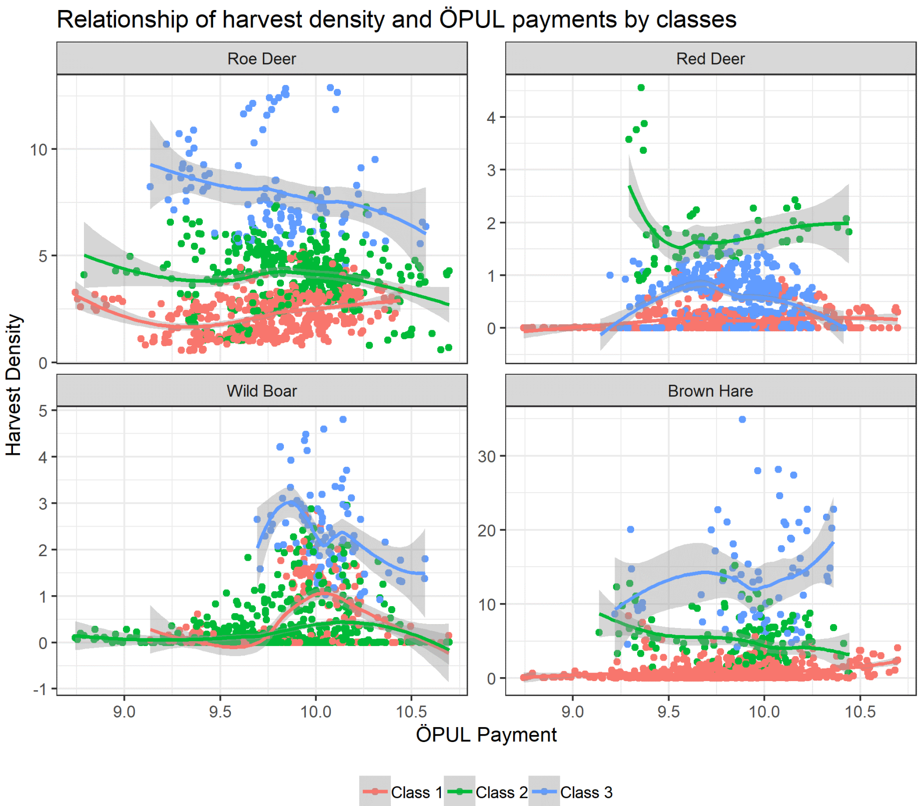

Figure 2.7: Relationship between the log of agri-environmental payments per km^2 and harvest densities of four wildlife species, separated by the latent classes. Note that the classes cannot be compared across different species, but estimated parameters are only significant in class 1 (red) for all species. Plots produced in ggplot2 (Wickham, 2016). Data sources: AWI, Statistik Austria.

Table 2.2: Estimation results of four latent class models regressing wildlife harvest on agri-environmental policy indicators and other district-level characteristics in Austria

	Roe Deer			Red Deer			Wild Boar			Brown Hare		
	coef	s.e.		coef	s.e.		coef	s.e.		coef	s.e.	
CLASS MEMBERSHIP MODEL												
intercept class1	-2.296	1.086	**	1.556	0.689	***	19.521	575.356		-6.827	2.827	
intercept class2	-2.748	1.155	**	-3.321	1.736		10.162	574.172		-1.969	2.481	
Altitude class1	4.303	1.741	**	-1.655	0.649	***	-11.28	5.986		17.875	6.693	
Altitude class2	4.044	1.77	**	0.907	1.111	***	7.81	3.631		6.922	6.33	
East class1	3.438	1.255	***	-0.934	0.63	***	-16.518	574.845		3.251	1.493	**
East class2	4.417	1.285	***	0.138	1.674	***	-12.666	574.222		0.985	1.346	***
South class1	1.863	1.174		0.501	0.618	***	-8.498	2445.02		14.878	665.275	
South class2	2.171	1.21	*	1.525	1.095	**	9.728	2460.266		-2.306	342.275	
OUTCOME MODEL												
intercept class2	2.345	0.847	***	9.103	1.565		1.69	1.345	*	-5.51	3.468	
intercept class3	-0.3	4.027		4.577	0.913		22.811	2.958		57.171	5.898	
log(Farm Premium) class1	0.617	0.143	***	-0.055	0.11	***	0.42	0.146	***	0.217	0.096	***
log(Farm Premium) class2	1.519	0.115	***	0.37	0.339	***	0.053	0.133	***	3.378	0.75	**
log(Farm Premium) class3	3.289	0.922	***	-0.54	0.157	***	0.722	0.286	***	-13.553	1.429	**
log(ÖPUL Payment) class1	0.723	0.206	***	0.781	0.199	***	-0.05	0.317	**	-0.028	0.176	***
log(ÖPUL Payment) class2	-0.278	0.241		-0.552	0.649		0.445	0.186		-1.258	0.478	
log(ÖPUL Payment) class3	0.016	0.466		0.319	0.286		-5.44	0.743		-1.956	0.591	
Average Minimum Temp	-0.124	0.035	***	-0.009	0.035	***	-0.159	0.035	***	0.073	0.035	***
Precipitation	0.011	0.003	***	0.008	0.003	***	-0.006	0.003	***	-0.001	0.003	***
Altitude	-1.077	0.164	***	0.252	0.175	***	-0.517	0.153	***	-0.36	0.146	***
Hunting Licenses	-0.062	0.02	***	-0.288	0.025	**	-0.005	0.019	***	-0.068	0.022	***
Grassland	0.031	0.01	***	-0.006	0.01	***	-0.026	0.01	***	0.026	0.011	***

	Roe Deer			Red Deer			Wild Boar			Brown Hare		
	coef	s.e.		coef	s.e.		coef	s.e.		coef	s.e.	
Grain Crops	0.122	0.024	***	-0.041	0.026	***	0.073	0.023	***	0.064	0.025	***
Sugar Beet	-0.721	0.065	***	0.437	0.085	***	0.739	0.073	***	0.028	0.081	***
Maize	0.149	0.031	***	-0.02	0.033	***	-0.078	0.031	***	-0.092	0.034	***
Tree Density	-0.963	0.334	***	3.587	0.416	**	-1.36	0.332	***	-2.591	0.375	**
Unproductive land	0.58	0.343	*	1.388	0.348	*	-0.521	0.346	*	-0.369	0.328	*
Average Farm Size	-0.041	0.012	***	0.021	0.013	***	0.036	0.012	***	-0.033	0.014	***
After 2006 (Dummy)	-0.372	0.109	***	0.374	0.109	***	0.756	0.109	***	-0.317	0.109	***
						LINK FUNCTION						
Linear 1 (intercept)	-0.841	0.651		-0.949	0.212	***	-0.717	0.405	**	4.431	1.506	
Linear 2 (std err)	0.796	0.02	***	0.233	0.006	***	0.31	0.008	***	1.987	0.049	***

Note: *p<0.1; **p<0.05; ***p<0.01
Source: own calculations

Table 2.3: Model statistics and class membership probabilities in four latent class models relating wildlife harvest density to agri-environmental subsidies

Model	# Obs	# Classes	loglik	# params	AIC	BIC	%class1	%class2	%class3
Roe Deer	841	3	-1076.6	30	2213.21	2289.508	48	40	12
Red Deer	841	3	-40.9703	30	141.9406	218.2395	51	6	43
Wild Boar	841	3	-242.053	30	544.1068	620.4056	20	70	10
Brown Hare	841	3	-1803.78	30	3667.563	3743.862	78	15	7

Source: own calculation

which means that traditionally intensive farmers generally benefit more (i.e. get higher payments) than extensive farmers. Roe deer has adapted exceptionally well to intensive agriculture, any may therefore benefit from additional food sources that could be provided through intensified input use caused by pillar 1 payments.

ÖPUL payments, on the other hand, are only significant and positive in class 1. Given the low harvest density in this class, ÖPUL payments could improve habitat conditions in areas where extensive agriculture in combination with high altitudes does not provide optimal feeding conditions, as it could help farmers to keep cultivating otherwise unprofitable land.

Table 2.4: Descriptive statistics by latent classes in the roe deer estimation

	Class1		Class2		Class3	
	Mean	SD	Mean	SD	Mean	SD
Roe deer harvest	2.252	0.903	3.992	1.246	7.894	2.032
Farm Premium	17772	8886	21885	8798	25553	5704
ÖPUL Payment	18364	5366	20709	6175	18890	6064
Min Avg. Temp	-2.882	1.811	-2.376	1.591	-2.283	1.112
Precipitation	86.467	21.338	78.098	19.656	81.406	10.068
Altitude	0.83	0.54	0.641	0.426	0.475	0.126
Hunting Licenses	2.016	2.506	1.942	1.911	1.765	0.61
Grassland	27.234	19.917	20.395	21.271	16.42	4.963
Grain Crops	6.963	8.473	10.664	9.148	8.886	4.391
Sugar Beet	0.336	0.84	0.819	1.329	0.453	0.433
Maize	2.115	2.012	2.739	1.845	3.536	1.546
Tree Density	44.736	15.179	46.091	16.283	44.658	13.126
Unproductive Land	0.135	0.176	0.07	0.133	0.021	0.011
Farm Size	30.267	11.274	31.327	13.349	23.808	4.526

Source: own calculations

Red deer

Again, ÖPUL payments are only significant effect for class 1. The highest harvest density of red deer (1.8 animals/km^2) is in class 2. As can be seen from Figure 2.6, class 2 is mainly restricted to a few mountaineous districts in the center and West of Austria. This class is characterized by the lowest farm premium by far, highest average altitude (1095 m), highest grassland (42 ha/farm), unproductive land (0.26 ha/farm), and tree density (52.7%), lowest grain crops (1 ha/farm), sugar beet (0.02 ha/farm) and maize (0.4 ha/farm). Class 3 has the second highest harvest density (0.6 animals/km^2) and also the second highest average altitude (791 m) and grassland (25.6 ha/farm). It is high in grain crops, sugar beet, and maize, and highest in average farm sizes (32.5 ha). Finally, class 1 with the lowest harvest density of red deer (0.15 animals/km^2) is characterized by the lowest altitude (594 m) and grassland (18.7 ha/farm), lowest average tree density (44%), highest in grain crops (9.5 ha/farm) and maize (2.9 ha/farm) and the highest farm premium (Table 2.5).

Clearly out of the three classes estimated for red deer, class 1 districts represent the most intensively used agricultural land. As the parameter for ÖPUL payments is positive and significant in this class, it suggests that the agri-environmental program could help red deer to thrive in particularly intensively used agricultural areas. This contrasts with our findings for roe deer, and suggests that red deer would particularly benefit from environmental services provided in areas that are less suitable for them (as the low harvest density suggests).

Table 2.5: Descriptive statistics by latent classes in the red deer estimation

	Class1		Class2		Class3	
	Mean	SD	Mean	SD	Mean	SD
Red deer harvest	0.15	0.2	1.848	0.697	0.58	0.42
Farm Premium	22230	8154	8847	4597	19948	9010
ÖPUL Payment	19900	6782	17818	5995	19055	4507
Min Avg. Temp	-2.287	1.388	-4.261	1.5	-2.732	1.841
Precipitation	80.723	20.333	97.149	15.803	82.115	19.219
Altitude	0.594	0.346	1.095	0.389	0.791	0.571
Hunting Licenses	1.564	0.976	0.888	0.581	2.597	2.945
Grassland	18.706	15.23	42.058	14.93	25.595	23.051
Grain Crops	9.471	7.788	1.048	1.45	9.017	9.515
Sugar Beet	0.555	1.024	0.016	0.036	0.633	1.175
Maize	2.91	1.763	0.403	0.456	2.42	2.076
Tree Density	44.107	15.103	52.714	16.576	45.65	15.366
Unproductive Land	0.069	0.141	0.267	0.218	0.1	0.138
Farm Size	27.787	9.329	30.368	10.741	32.541	14.13

Source: own calculations

Wild boar

The increase in wild boar populations in Austria is a relatively recent phenomenon. While wild boar has occurred in viable population sizes throughout history, it has been hunted to extinction in many areas due to its detrimental effects on agricultural harvest. Today, wild boar reaches its highest population densities in the eastern parts of Austria, although the occurrence has increased in central and western districts in recent years. This fact of the population history needs to be taken into account when interpreting the results of wild boar.

The highest harvest density of wild boar is found in districts of class 3, which are second-lowest in average altitude (321 m) and tree density (40%) and lowest in annual precipitation, grassland (6.3 ha/farm). They are second-highest in maize, sugar beet, and highest in average farm size. Class 1 is similar to class 3, but has an even lower tree density (29%) and average altitude (311 m), higher maize (4.3 ha/farm), sugar beet (1.7 ha/farm) and grain crops (17.3 ha/farm) and precipitation. It also has a substantially lower harvest density of wild boar (0.6 animals/km^2). Class 2, with the lowest harvest density of wild boar (0.3 animals/ha), represents the most extensive agricultural class. Average altitude is highest (884 m) and so is grassland (29.8 ha/farm), while grain crops, maize, and sugar beet, and average farm size are lowest (Table 2.6).

Our results suggest that harvest densities of wild boar are highest in medium-to high intensity farmed areas (class 3), while they are much lower in the other two (high-intensity and low-intensity) classes. But only class 1, the high-intensity farming class, has a significantly negative effect of ÖPUL payments on the harvest density of wild boar. We can spectulate that in these high-intensity regions, limiting the amount of fertilizer being used could actually reduce the outputs and therefore feeding sources of wild boar. This contrasts with class 2, where intensity is not as high and therefore the participation in ÖPUL may not or only marginally limit input use. This explanation would be consistent with our theoretical model.

Table 2.6: Descriptive statistics by latent classes in the wild boar estimation

	Class1		Class2		Class3	
	Mean	SD	Mean	SD	Mean	SD
Wild Boar Harvest	0.573	0.715	0.255	0.492	2.353	0.915
Farm Premium	27675	7056	17743	8263	23939	7262
ÖPUL Payment	20238	6506	18577	5647	23493	4170
Min Avg. Temp	-1.446	1.243	-3.098	1.589	-1.535	1.263
Precipitation	70.121	13.379	88.81	19.203	62.996	9.758
Altitude	0.311	0.119	0.884	0.472	0.321	0.089
Hunting Licenses	3.271	3.05	1.541	1.685	2.082	0.999
Grassland	8.909	7.203	29.754	20.072	6.258	3.486
Grain Crops	17.327	6.935	5.07	6.362	16.372	7.091
Sugar Beet	1.673	1.41	0.248	0.747	0.334	0.354
Maize	4.288	1.158	1.921	1.796	3.257	1.817
Tree Density	29.01	13.311	50.952	12.868	40.05	8.208
Unproductive Land	0.019	0.019	0.121	0.169	0.064	0.134
Farm Size	33.297	8.345	28.445	12.623	33.353	10.711

Source: own calculations

Brown Hare

Brown hare is generally seen as an indicator species of environmental quality and has been used in the evaluation of AEPs before (Ujhegyi et al., 2015). Harvest numbers of brown hare have gone down in Austria over the last 30 years, as was shown in Figure 2.3. We find the highest harvest density of brown hare in class 3 (13.4 hares/km^2), which is characterized by the lowest average altitude (277 m) and precipitation levels, highest amount of grain crops (19 ha/farm), low tree density (22%), and large farm sizes (35 ha). In addition, class 3 farms receive the highest farm premium and ÖPUL payments. Class 2 is similar to class 3, but has substantially lower harvest densities of brown hare (5 animals/km2), higher amounts of maize (4.2 ha/farm) and higher tree density (34%) and a slightly higher average altitude (320 m). Finally, class 1 has the lowest brown hare harvest densities, and consists mainly of the hilly to mountaineous central, western, and southern Austrian districts. Therefore, it is highest in average altitude (830 m), tree density (49.7%), and grassland (27.6 ha/farm), has the smallest average farm size (27.7 ha) (Table 2.7).

Clearly, highest densities of brown hare are found in the most intensively used agricultural areas in Austria. Here, payments for more extensive farming methods do not change hare populations significantly. Hares in areas dominated by extensive farming, however, may suffer from additional constraints when farming methods are extensified in company with the agri-environmental subsidy. This stands in contrast with previous findings on the decline of brown hare (Bauer, 2001).

Table 2.7: Descriptive statistics by latent classes in the brown hare estimation

Variable	Class1		Class2		Class3	
	Mean	SD	Mean	SD	Mean	SD
Brown Hare Harvest	0.741	1.06	4.989	3.001	13.399	6.776
Farm Premium	17900	8471	28857	4240	28701	3360
ÖPUL Payment	19051	6048	20040	5175	21762	5181
Min Avg. Temp	-2.959	1.607	-1.39	1.278	-1.426	1.257
Precipitation	86.259	19.829	71.298	13.828	65.509	14.118
Altitude	0.83	0.479	0.32	0.095	0.277	0.093
Hunting Licenses	1.635	1.671	3.461	3.397	2.125	1.27
Grassland	27.637	20.051	8.827	8.172	6.159	6.36
Grain Crops	6	6.806	17.145	7.576	19.282	7.024
Sugar Beet	0.182	0.481	1.822	1.605	1.707	1.199
Maize	2.112	1.924	4.176	1.006	3.611	1.398
Tree Density	49.655	13.668	34.957	10.795	22.668	8.823
Unproductive Land	0.117	0.168	0.016	0.009	0.024	0.029
Farm Size	28.71	12.418	33.587	9.012	34.938	8.092

Source: own calculations

2.7 Discussion and Policy Implications

Our analysis highlights that population densities of four distinct wildlife species depend on a variety of factors, both natural and anthropogenic. Regarding the natural factors, we find that roe deer, wild boar and brown hare have their highest densities in low-altitude regions, while high densities of red deer are found in the mountaineous districts of Austria. Particularly roe deer and wild boar thrive in intensively used agricultural areas dominated by grain crop farming, lower forest densities, and lower amounts of grassland. Red deer, on the other hand, are mostly found in districts with high forest densities and generally less-intensively used agriculture.

The latent class estimation has mainly separated districts into more and less agriculturally intensive districts for all species. It has also revealed that ÖPUL payments may have positive or negative population effects, depending on the species and the intensity of farming. While the three ungulates in our case study are certainly not threatened and may not have been relevant in the current policy design, our results suggest that agri-environmental programs designed to enhance biodiversity may have unintended side effects. For example, when red and roe deer populations are increased due to an AEP, problems with forest damage and or wildlife-vehicle-collisions could be exacerbated.

It must be recognized that the non-response of a wildlife species to the adoption of an AEP can have two sources: first, there may be no behavioral change of the farmer due to adverse selection. Second, even if there is a behavioral change, it may not cause a change in the habitat or food availability of the given species. The ecological literature has viewed habitat requirements of a species through the lens of Liebig's law of the minimum (Krebs, 2013), where only a change in the limiting factor within the habitat will cause a population change. Future research could try to separate the farmer behavior effect from the ecological effect of a limiting factor by explicitly incorporating farmer behavior into the model, if more detailed data become available in the future (e.g. through remote sensing). Nevertheless, given that the goal of agri-environmental policy is to improve environmental quality and enhance biodiversity, zero-outcomes will produce deadweight losses of subsidy either way.

A key question for the future development of agri-environmental programs is whether the conditions for participation should be the same for all farmers within given administrative boundaries (e.g. country or state borders). As we explain in our theoretical model, farmers on low-productivity land may be able to reap the full benefit of the subsidy without improving environmental conditions. On the one hand, it has been argued that the conservation of marginal farmland could improve biodiversity and that abandoning farming in these areas could severely threaten the populations of certain (endangered) species. In this case, the payment is justified by preventing environmental degradation for certain species. This perspective is supported by Halada, Evans, Romão, and Petersen (2011), who found that 63 out of 198 habitat types defined in Natura 2000 conservation policy benefit from agricultural activities. However, some scholars have argued that biodiversity values are often higher in land were farming is abandoned and where the landscape is transformed by natural succession (Merckx & Pereira, 2015). Given this perspective, the agri-environmental payment to marginal farmers not only produces dead-weight losses, but it may actually be counterproductive for reaching biodiversity goals.

The effect of an AEP may also be ambiguous due to reasons that farmers have no control over. For example, forest cover will to a large part be the result of infrastructure development and zoning policies rather than a farmer's production decisions. Nevertheless, currently the participation in an AEP lies strictly in the hands of the farmer, whether or not a gain in environmental quality or other public goods is likely. We propose the following succession of steps to guide the design of future AEPs. (1) Identify target species to be protected by the AEPs and (2) identify the corresponding habitats. Then (3) design AEPs with a clear ecological focus in mind. That is, farmers can only participate in an AEP if the regional habitat characteristics provide suitable habitat conditions for a species in question. In effect, this is a call for a regionalization of agri-environmental policy. Instead of broad measures that have, as our research has shown, questionable and ambiguous effects on wildlife species, only farmers in a specific region can participate in a program that targets certain species of conservation or other (e.g. hunting, forest protection, etc.) interest.

Using the approach outlined above could (1) help to focus agri-environmental policy goals towards measurable impacts, (2) increase the efficiency of spending in AEPs by reducing the dead-weight losses, and (3) reduce complexity and uncertainties associated with purely outcome-based payment models. It thereby presents a compromise between the status quo (unrestricted access to payments) and the possibly ecologically superior, but technically often infeasible outcome-based renumeration.

It must be pointed out that the scale of this study is relatively coarse. Given finer resolution data (e.g. municipality level or below) of annual game harvest, one could try to study the impact of agricultural policy at the measure level, e.g. by separating the impacts of catch crops and organic farming. More detailed data may become available in the future as hunting associations modernize their data collection and monitoring capabilities.

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Chapter 3

The Impact of Natura 2000 Designation on Agricultural Land Rental Prices in Germany

Dieter Koemle, Sebastian Lakner and Xiaohua Yu[1]

Abstract

Designation of Natura 2000 areas has been a major cornerstone in the EU's biodiversity policy. However, it has also triggered resistance from land users due to increased regulations on land use and related value change. This study first builds up a theoretical model for rent change due to land regulation, and then empirically investigates whether farmland rents in Germany are affected by Natura 2000 designation. Because Natura 2000 designation and rental prices are likely endogenous, we use the matching procedure by Imbens and Hirano (2004) based on a zero-inflated beta generalized propensity score on German district level agricultural census data. Our results suggest that overall, rental prices of grassland, arable land, and on average are affected negatively by Natura 2000 designation.

Key words: Natura 2000, agricultural land rent, Germany, generalized propensity score, zero-inflated beta model

3.1 Introduction

Regulations on land use and farming practice could change the land value. In order to reduce biodiversity loss in modern agro-ecosystems, the EU has introduced regulations to integrate the goals of the Bern Convention on Biodiversity into agricultural policy. Recent policy measures include the cross compliance and greening of Pillar 1 direct payments (Ciaian, Kancs, & Swinnen, 2012; Ciaian, Kancs, & Swinnen, 2014; Feichtinger & Salhofer, 2016; Pe'er et al., 2017), voluntary agri-environmental programs (Batáry, Báldi, Kleijn, & Tscharntke, 2011; Besnard & Secondi, 2014; Keenleyside, Beaufoy, Tucker, & Jones, 2014; Kilian, Antón, Salhofer, & Röder, 2012), and the establishment of conservation strategies including financial compensations for extensive

[1] The paper was written by DK. The idea was jointly developed by DK and XY. SL provided valuable comments on content and helped DK with data collection. Data were analyzed by DK. XY provided comments on methodology.

farming practices in environmentally sensitive areas (Olmeda, Keenleyside, Tucker, & Underwood, 2014). A central instrument for biodiversity protection and enhancement is the Natura 2000 network of protected areas throughout Europe. Natura 2000 claims to be the largest international network of protected sites[2] in the world, with 18% of the total EU land area and 6% of the EU's marine territory being set under Natura 2000 designation. Land designated to Natura 2000 plays a key role in ensuring the goals of the habitats and birds directives are met, so that "all habitats of community interest are maintained or restored to Favourable Conservation Status" (Gantioler et al., 2013; Olmeda et al., 2014). Once a site is designated, member states are required to manage and protect it in accordance with the terms of Article 6 of the habitats directive (Commission, 2014).

Annexes I and II of the Habitats Directive respectively define the habitat types and the species intended for protection. Of the 198 habitat types specified by Annex I of the habitats directive, 63 have been found to depend on or profit from agricultural activities (Halada, Evans, Romão, & Petersen, 2011). Twenty-eight habitat types can be threatened by the abandonment of low intensity agriculture (Ostermann, 1998). With the extension of the Natura 2000 network, policy makers are faced with trading off the interests of conservationists against other types of land users, particularly farmers (Geitzenauer, Hogl, & Weiss, 2016). While some EU countries have designated sufficient areas as Natura 2000 sites, others have been mandated by the European Commission to nominate additional sites.

Besides its ecological impacts, the designation of Natura 2000 sites may also considerably alter economic conditions for land users. Policies related to land use may have a particularly strong impact on land prices due to the low supply elasticity of land (Floyd, 1965). For example, the CAP (Common Agricultural Policy) direct payments consisting of coupled, decoupled, and environmental payments, theoretically may increase land prices considerably (Feichtinger & Salhofer, 2016; Kilian et al., 2012; Klaiber, Salhofer, & Thompson, 2017; Michalek, Ciaian, & Kancs, 2014), particularly when there is a surplus of entitlements[3] (Ciaian et al., 2014). However, the empirical evidence is mixed, and other authors find little or no direct relationship between land prices and various forms of direct payments (Guastella, Moro, Sckokai, & Veneziani, 2014). Ciaian et al. (2012), Ciaian et al. (2014) present a conceptual model explaining how cross compliance measures reduce farmers' total benefits from subsidies and therefore the capitalization of the pillar 1 payments into land values. Kilian et al. (2012) confirm findings by Goodwin, Mishra, and Ortalo-Magné (2003) that subsidies for agri-environmental programs may not or even negatively affect land rents, as farmers face additional costs to keep up higher environmental standards. Land subject to Natura 2000 designation is automatically subject to the rule of no deterioration (Art. 6(2) of the Habitats Directive), and therefore may decrease farmers' flexibility in input use. A suboptimal input mix will necessarily lead to profit losses if imposed production constraints are not sufficiently compensated. Letort and Temesgen (2014) provide evidence of decreases in land prices in the Bretagne region, France, for water protection policies.

[2] Natura 2000 sites include Special Protection Areas (SPAs) according to the Birds Directive (79/409/EEC) and Special Areas of Conservation (SACs) according to the Habitats Directive (92/43/EEC)

[3] To receive Pillar 1 direct payments for one hectare of land, farmers need entitlements for that hectare. Surplus of entitlements means that a farmer has more entitlements than hectares.

According to Corine Land Cover (CLC) 2012 data, farmland accounts for 34% of total Natura 2000 land area in Germany. While agricultural land has been declining, the percentage of forests inside Natura 2000 sites has increased over the time-frame 1990-2012. Because extensive livestock management and other low-intensity farming practices required by Natura 2000 have become unprofitable, key farmland habitats and species of community interest are under pressure. Germany has therefore picked up the EU's offer to subsidize farmers in designated sites through the rural development fund (pillar 2) of the Common Agricultural Policy.

The land use change under the Natura 2000 regulation might be linked to land value change. It is important to understand how the change impacts the land value, as this is related to the effectiveness of the economic compensation in the policy. By using regional aggregate data and a generalized propensity score matching procedure, we empirically study the link between land rental prices and the share of farmland on Natura 2000 sites. While others have studied the effects of subsidies in general (Feichtinger & Salhofer, 2013), agri-environmental programs (Goodwin et al., 2003; Kilian et al., 2012), or water conservation policies (Letort & Temesgen, 2014) on land prices, we specifically investigate the impact of a European conservation policy on land rental prices at the aggregate level. Knowledge about this relationship is important because Natura 2000 designation could affect many farmers across Europe. We specifically analyze the impact of Natura 2000 on farmland rents for average rent, grassland rent, and arable land rent separately.

3.2 Background

Germany has a total of 5206 designated Natura 2000 sites, 4557 of which are SACs (Special Areas of Conservation according to the Habitats Directive) and 742 are SPAs (Special Protection Areas according to the Birds Directive). Combined, they cover 15.4% of the terrestrial area and 45% of marine areas. Although EU countries were supposed to report designated sites to the EU Commission three years after the Habitats Directive went into force in 1992, by 1995 Germany had not reported a single site. Following a series of legal claims at the European Court of Justice, Germany reported sufficient Natura 2000 sites by 2005, and the Commission dropped any legal claims that were still pending. Well-known Natura 2000 sites in Germany include the Harz Mountains, the Lüneburg Heath, and the Black Forest in the South.

Problems with delayed implementation of Natura 2000 sites are not specific to Germany. A summary of several implementation difficulties is provided by Geitzenauer et al. (2016), who summarized implementation processes according to (a) institutional capacities, (b) the pressure for instutional change, and (c) the role of actors (authorities, environmental NGOs, and landowners). Implementation was slow not only due to lack of funding and lack of data, but also because farmers and landowners protested designation, as many of them feared losses in land values and production constraints. Local stakeholders were also concerned regarding the lack of involvement of local interest groups in the designation process (Geitzenauer et al., 2016). As Rauschmayer, van den Hove, and Koetz (2009) point out, participation in German site designation was largely restricted to public consultation processes, or to gain information on specific sites. In most cases, however, site designation was based on a top-down technocratic approach. On the other hand, in cases where public participation was encouraged, researchers noticed "participation fatigue" of stakeholders who stopped participating

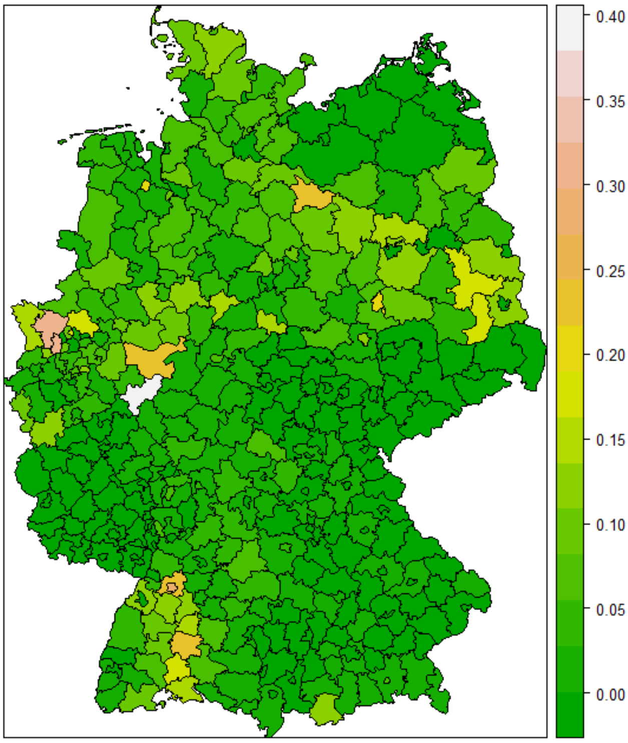

Figure 3.1: Share of farms receiving funding for Natura 2000 farming in German districts in 2010

after finding that their expertise was not considered in the designation process (Sauer, Luz, Suda, & Weiland, 2005).

Regulations for Natura 2000 sites enter agricultural policy through both pillars of the Common Agricultural Policy (CAP). First, cross compliance (CC) standards are usually higher in Natura 2000 areas than outside . Second, Article 38 of Council Regulation (EC) No 1698/2005 (European Agricultural Fund for Rural Development, EAFRD) establishes the framework for compensation of farmers directly affected by the Birds Directive and the Habitats Directive. Payments for conservation in sensitive areas may be channeled through voluntary agri-environmental programs (AEPs; EAFRD Code 214). However, these payments may be reduced inside protected areas in case requirements overlap with elevated CC requirements. Third, land that is protected by national and regional protective measures (e.g. "Natuschutzgebietsverordnung") can be additionally funded through the second pillar measure "Natura 2000 payments" (EAFRD Code 213) (Reiter & Sander, 2010). Participating farmers are to refrain from removing landscape elements and breeding sites of bird species, and they need permissions when implementing changes in the terrain or measures that affect the water balance. Further, farmers may be affected by hunting regulations and restrictions in planting non-local plants.

Natura 2000 designation and its integration into rural development programs are handled by the German states ("Bundesländer"). Therefore, it is up to the states whether they grant subsidies to farmers in Natura 2000 areas or not. For the programming period of 2007-2013, six states chose not to pay any subsidies for EAFRD Code 213. One of these six states, Saxony, did provide additional payments for Natura 2000 farming via the agri-environmental program Code 214. Also, participation in subsidized Natura 2000 farming was voluntary in the states of Saxony and Bavaria, while it was mandatory in all other states whether subsidies were paid or not. An additional complication of Bavaria was the substantial overlap between 213 and 214 measures, making it difficult to identify which farms or regions received money for farming in Natura 2000 areas. The remaining ten states paid a total of 119 million € in subsidies via Code 213, with the maximum amount of 31.2 € million paid by Brandenburg and Berlin combined, and the lowest amount by Hesse (2.1 million €) (Table 1). Payments were primarily focused on grasslands, but some states also subsidized extensively managed arable land. In the state of Lower Saxony, payments were only granted if the farmed land was protected according to state legislation in addition to Natura 2000 (Reiter & Sander, 2010; Tietz et al., 2007). In the state of Schleswig-Holstein, all Natura 2000 agricultural areas were subsidized with 80 € per hectare per year, while most other states had differentiated payments depending on the severity of the production impairment. Figure 3.1 shows where farms receiving payments in Natura 2000 are concentrated in Germany. Of particular notice is the Schwarzwald (black forest) in the South-West, the Lüneburg Heath and wetlands along the river Elbe to the south of Hamburg, protected areas in the Ruhrpott region, as well as protected areas to the east of Berlin.

3.3 Theoretical framework

3.3.1 Basic model

In the classical economic literature, without regulation, land is assumed to be freely used to maximize its profit. Land market values therefore reflect the marginal revenue of production. Regulating the use of land through policies such as environmental zoning (e.g. Natura 2000), however, would change the rent due to less choice, even though there is some compensation. Farmland rent r enters a farmer's profit function π as an input cost on land L:

$$\pi = f\left(L, N\left(L\right)\right) + v\left(N\left(L\right)\right) - \sum_i w_i x_i\left(N\left(L\right)\right) - rL \tag{3.1}$$

where $f\left(L,\ N\left(L\right)\right)$ is total revenue from production and non-Natura 2000 related subsidies, v is the subsidy for Natura 2000 land and N is an indicator of Natura 2000 farmland, x_i are other inputs and w_i is the marginal cost of x_i. We assume that input costs are exogenous, but input use is related to Natura 2000 farmland. The farmer maximizes profit by setting the marginal profit of each input to zero. Partially differentiating the profit function with respect to land and setting zero yields

$$r = \frac{\partial f\left(L,\ N\right)}{\partial L} + \frac{\partial f\left(L,\ N\right)}{\partial N}\frac{\partial N}{\partial L} + \frac{\partial v}{\partial N}\frac{\partial N}{\partial L} - \sum_i w_i\left(\frac{\partial x_i}{\partial N}\frac{\partial N}{\partial L}\right)$$

or

$$r = \frac{\partial f\left(L,\ N\right)}{\partial L} + \left[\frac{\partial f\left(L,\ N\right)}{\partial N} + \frac{\partial v}{\partial N} - \sum_i \frac{\partial x_i}{\partial N}w_i\right]\frac{\partial N}{\partial L} \tag{3.2}$$

Table 3.1: Summary of EAFRD Code 213 payments in the German States for farming in Natura 2000 areas for the 2007-2013 programming period (Sources: State Rural Development Programs and their evaluation reports)

State[4]	Budget[5]	Area	Farms	Details	Premium
Baden-Württemberg	€ 7.91 Mio	8385 ha	3043	• Extensive use of semi-natural dry grasslands (Code 6210), species-rich Nardus grasslands (Code 6230) and milinia meadows on calcareous, peaty and clayey-silt-laden soils (Code 6410) • Extensive use (grazing) of European dry heaths (Code 4030), Juniperus communis formations on heaths or calcareous grasslands (Code 5130), Xeric sand calcareous grasslands (Code 6120), of semi-natural dry grasslands (Code 6210), and species-rich Nardus grasslands (Code 6230)	50-200 €/ha
Bavaria	€ 2.60 Mio	883 ha	338	• Voluntary participation • Restrictions in the application of mineral fertilizers and chemical pesticides on arable land • grassland – extensive mowing – restriction on mineral fertilizers and chemical pesticides – restriction on organic fertilizers excluding solid dung • ponds	85-470 €/ha

[4] The states of Hamburg, Mecklenburg-Vorpommern, Rheinland-Pfalz, and Thuringia did not participate in EAFRD Code 213. The state of Saxony paid Natura 2000 subsidies for voluntary farmer participation via the agri-environmental programs EAFRD Code 214

[5] Budget for the entire programming period 2007-2013 according to ex-post evaluation reports

State	Budget	Area	Farms	Details	Premium
Brandenburg/ Berlin	€ 31.20 Mio	38819 ha	594	• Grassland – Extensive use of grassland – Late and constrained use of grassland – Conservation of wetlands • Extensive production methods on arable land	30-200 €/ha
Hesse	€ 2.10 Mio	3904 ha	509	• Only on grassland • No chemical-synthetic pesticides and fertilizers • No irrigation and land development • Conduct agricultural use at least once a year • Other regulations regarding time of mowing etc.	200 €/ha
Lower Saxony and Bremen	€ 17.57 Mio	21056 ha	1826	• Payment levels are based on the production handicap • Constraints – No machine tillage from March 1 to June 15 – No conversion of grassland to arable land – no fertilization – 2.5m grassy margins	33-874.5 €/ha

State	Budget	Area	Farms	Details	Premium
Northrhine-Westphalia	€ 21.20 Mio	35349 ha	5316	• Payments for permanent grasslands inside SACs and SPAs • Payment level and constraints depending on protection level (high, medium, low) • Constraints – No conversion of grasslands to arable land – No drainage – No removing of biotopes and and other habitat features – Protection of relief features	36-98 €/ha
Saxony-Anhalt	€ 23.29 Mio	27217 ha	444	• No fertilization on grassland • constraints on the use of fertilizers, pesticides, and tillage methods • Hamster protection	8-199 €/ha
Schleswig-Holstein	€ 13.3 Mio	18277 ha	1196	• No deep tillage on grasslands • No drainage • No removal of traditional "Beet-Graben- Systems"	80 or 150 €/ha

Rent is therefore described by two components, the marginal revenue of land $\frac{\partial f(L,\ N)}{\partial L} > 0$, and the effect of Natura 2000, $\left[\frac{\partial f(L,\ N)}{\partial N} + \frac{\partial v}{\partial N} - \sum_i \frac{\partial x_i}{\partial N} w_i\right] \frac{\partial N}{\partial L}$.

$\frac{\partial f(L,\ N)}{\partial N} < 0$: it is plausible that Natura 2000 designation reduces the marginal productivity and therefore marginal revenue of land, given unchanged output prices. We name this effect the revenue effect of Natura 2000.

$\frac{\partial v}{\partial N}$ will likely be positive. We name this the subsidy effect of Natura 2000. As v is the total payment per farm, a positive sign would mean that the per-farm payment increases as designated farmland increases.

The sign of $\frac{\partial x_i}{\partial N}$ will depend on the specific input. For example, it could be negative regarding the use of synthetic fertilizers, while it may be positive for labor. We call this term the input effect.

Finally, the sign of $\frac{\partial N}{\partial L}$ is likely to be positive. This means that a larger farm size is associated with a higher share of Natura 2000 farmland. Intuitively, if a farmer owns more land, the chance of owning some land on protected land should also be higher.

The overall effect of Natura 2000 at the farm level will therefore depend on whether the sign of the aggregate components in the brackets of equation (3) is positive or negative. According to the arguments outlined above, it will be negative as long as reductions in input costs do not outweigh the (negative) productivity and subsidy effects. It will be zero, if the effects compensate each other exactly.

3.3.2 The effect of Natura 2000 designation on average rent

In the empirical part of this paper, we study the effect of Natura 2000 designation on district average rent. From our derivation above, the average rent within a district is

$$\frac{1}{F}\sum_j r_j = \frac{1}{F}\sum_j \left[\frac{\partial f_j\,(L,\ N)}{\partial L} + \left(\frac{\partial f_j\,(L,\ N)}{\partial N} + \frac{\partial v_j}{\partial N} - \sum_i \frac{\partial x_{ij}}{\partial N} w_{ij}\right)\frac{\partial N}{\partial L}\right] \tag{3.3}$$

where j indexes the individual farms and F is the total number of farms. We can now decompose (4) explicitly into Natura 2000 farms (FN) and non-Natura 2000 farms (FO).

$$\begin{aligned}\frac{1}{F}\sum_j r_j =& \frac{FN}{F}\left\{\frac{1}{FN}\sum_j \left[\frac{\partial f_j^N(L,N)}{\partial L} + \right.\right.\\ & \left.\left.\left(\frac{\partial f_j^N(L,N)}{\partial N} + \frac{\partial v_j^N}{\partial N} - \sum_i \frac{\partial x_{ij}^N}{\partial N} w_{ij}\right)\frac{\partial N}{\partial L}\right]\right\} + \\ & \frac{FO}{F}\left\{\frac{1}{FN}\sum_j \frac{1}{FO}\left[\frac{\partial f_j^O(L,N)}{\partial L} + \frac{\partial f_j^O(L,N)}{\partial N}\frac{\partial N}{\partial L}\right]\right\}\end{aligned}$$

where $\frac{\partial f_j^O(L,\ N)}{\partial N}\frac{\partial N}{\partial L}$ is the revenue spillover effect of Natura 2000 designation on those farms that are not in any designated site. For example, it could be the increased competition for non-designated land, which could lead to an increase in farmland rents. It would be negative if other components of revenue, e.g. other second pillar payments, are reduced due to more funding going to Natura 2000 farmers. We assume that other input costs for non-Natura 2000 farms remain unchanged.

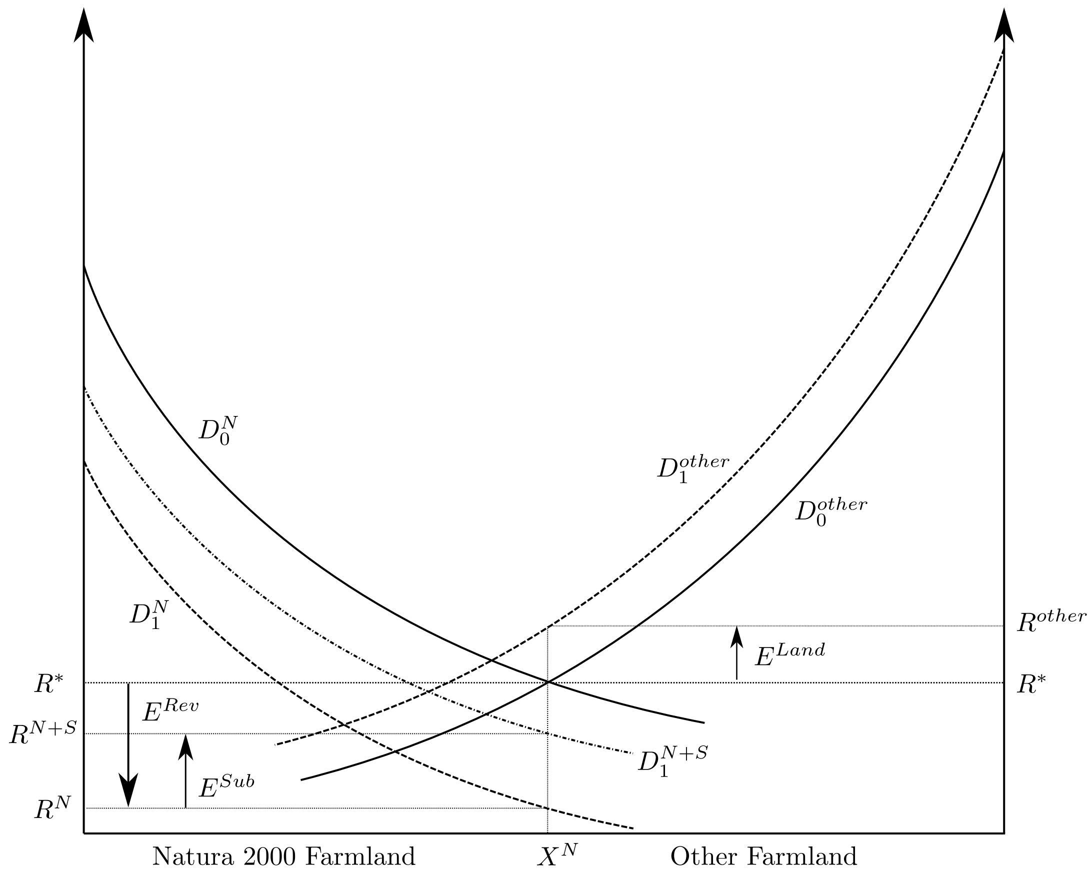

Figure 3.2: The impact of Natura 2000 designation on farmland rental prices inside and outside Natura 2000 sites

To sum up, Natura 2000 farming could have a positive, as well as a negative effect on average land rental prices, and the sign depends on (1) the relationship between revenues, subsidies, and input adjustment of Natura 2000 farmers, as well as (2) possible spillover effects of Natura 2000 designation to non-Natura 2000 farmers.

To make the above descriptions more tractable, the mechanism is explained more clearly in Figure 3.2. Similar to Michalek et al. (2014), the horizontal axis shows the total amount of farmland. From the left to X^N is farmland under Natura 2000, and from the right side to X^N is the amount of farmland without Natura 2000. On the vertical axes we show land rental price and subsidy. D_0^N and D_0^{other} respectively represent the demand curves for Natura 2000 and non-Natura 2000 (other) farmland *before* designation, and the rental price is R^*. After designation, the demand curve for Natura 2000 farmland could shift down to D_1^N, given the revenue effect E^{Rev}. If a subsidy for Natura 2000 farming is granted, the demand curve will shift up to D_1^{N+S}, the corresponding effect being E^{Sub} in Figure 3.2. Non-Natura 2000 farmland now becomes more scarce, which could lead to an upward shift in its demand function (D_1^{other}). The new average price is then a weighted average between R^N, R^{N+S}, and R^{other}. Depending on the ratio of Natura 2000 farmers and other farmers, it may be lower or higher than the original equilibrium price R^*.

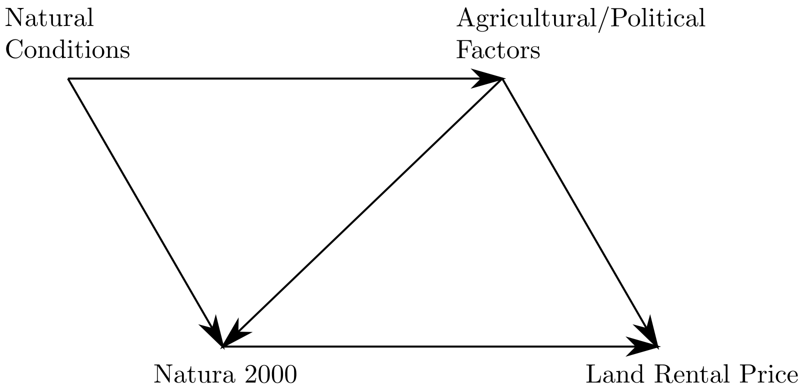

Figure 3.3: Graph of the confounding relationships between natural conditions, agricultural factors, and Natura 2000 designation and their impact on land price

3.4 Econometric models

Assignment to Natura 2000 farming may not be random. As described above, the designation process follows several steps including local, national and EU levels of policy making. Nevertheless, the first step in Natura 2000 designation is the environmental quality of a potential site which houses habitats and species of community interest. Natural conditions will also shape the agricultural and political environment of a region, which in turn influences site designation. The decision whether to subsidize Natura 2000 farming through 2nd pillar payments is made at the state level. It is therefore likely that the effect of Natura 2000 farmland on land rental prices is confounded with the effect of natural, agricultural and political factors as is shown in Figure 3.3. The analysis of the impact of Natura 2000 farming policy must therefore account for (1) the higher level political process of the Natura 2000 implementation strategy (i.e. the outcome of the rural development strategy), and (2) the actual outcome of the policy implementation (i.e. the number of Natura 2000 farms), and (3) the impact of designation on land rental prices. Therefore, if the impact of Natura 2000 farming on land prices is confounded with natural conditions and political and agricultural factors, standard regression analysis is not applicable and will produce biased results. A common method of controlling for confounding effects (i.e. to block the back-door paths shown in Figure 3.3) is to use the propensity score.

With binary treatments, the model of choice for the propensity score is usually a probit or logit. For continuous treatments, Hirano and Imbens (2004) provide an estimation procedure based on the *generalized propensity score*. The Hirano and Imbens estimator requires weak unconfoundedness between the treatment and the outcome variable, given all observed explanatory variables. The generalized propensity score is defined as $r(t, x) = f_{T|X}(t|x)$, which describes the conditional density of treatment t given covariates x.

Given our district-level data, the natural treatment variable that is provided by the agricultural census is the share of farms receiving Natura 2000 related subsidies. This treatment variable has two complications compared to a standard continuous treatment, in that (1) is a proportion (i.e. between zero and one), and that (2) about one third of the observations is zero due to the reasons described above. A less-elegant method would be to use a binary model to estimate the propensity score based on the presence of absence of Natura 2000 farming. More elegantly, the zero-inflated beta model (Ospina & Ferrari, 2010, 2012a, 2012b) can work with the properties of our data directly. The density function of the Beta distribution is a function of two parameters, μ with $0 < \mu < 1$, and $\phi > 0$:

$$f(N;\mu,\phi) = \frac{\Gamma(\phi)}{\Gamma(\mu\phi)\,\Gamma((1-\mu)\,\phi)} N^{\mu\phi-1}(1-N)^{(1-\mu)\phi-1}\,,\ N\ \in(0,1) \tag{3.4}$$

which is defined on the open interval $(0,1)$. μ denotes the expected value of the distribution N is the share of Natura 2000 farms, and ϕ is the precision parameter of the Beta distribution. Because the Beta distribution cannot be used to model proportional data that include zeros, Ospina and Ferrari propose a mixture of two models, in particular

$$bi_0(N;\alpha,\mu,\phi) = \begin{cases} \alpha & if\ N = 0 \\ (1-\alpha) f(N;\mu,\phi) & if\ N \in (0,1) \end{cases} \tag{3.5}$$

where α is the probability density in case $N = 0$. Therefore, bi_0 models the conditional distribution of Natura 2000 farming in a district, given any covariates. The zero inflated Beta model has three parameters that can be modeled separately. First, α is modeled as a binary model, e.g. a logit. Similarly, μ is also modeled using a logit specification. Finally ϕ is modeled using a log transformation of a linear model, which ensures positivity. Each partial model in the zero inflated Beta model can be defined separately, however, for consistency, we use the same covariates in the logit models and assume that the precision parameter is constant.

Imbens and Hirano (2004) stress that the overlap conditions given the covariates have to be maintained in order to produce reliable estimates. Schafer (2015) implements a procedure in `R` named `overlap_fun()` that balances the covariate overlap within the dataset based on the generalized propensity score. We adapted this procedure to work with the zero inflated Beta estimation function written by Ospina (Ospina & Ferrari, 2010) and included in the `gamlss` package (Rigby & Stasinopoulos, 2005; Stasinopoulos, Rigby, Heller, Voudouris, & Bastiani, 2017). To check for covariate balance after matching, we adopted the procedure of Imbens and Hirano (2004) by discretizing on three treatment groups and five GPS groups, to see whether there were significant differences between three classes of treatment. Table 3.2 presents the t-statistics of comparisons among three treatment groups after applying the matching procedure. In particular, each t-statistic tests whether the mean difference of the variable in question in one treatment group vs, the combination of the two other treatment groups is significantly different from zero. Only two t-statistics are significant at the 5% level (share of agricultural GDP in 1999, and average farm size), which makes us optimistic that the matching procedure based on the generalized propensity score has worked reasonably well.

The two separate parts of the zero inflated beta model also have an economic meaning related to the three stages outlined above. The binary logit model α describes the higher-level decision of a state of how to implement Natura 2000 farming policy (e. g. whether it should be subsidized or not). The second stage then models the actual outcome (i.e. the proportion of affected farmers), given the first stage.

The third stage is estimated after estimating the GPS and confirming its balancing property, which is the estimation of the impact of Natura 2000 farming on land rental

Table 3.2: T-statistics comparing three treatment level groups after matching

	Group 1	Group 2	Group 3
Median Share of Natura 2000 farming	0	0.008	0.072
Covariates		t-statistics	
Average Rent 1999	0.224	-1.111	-0.181
Grassland Rent 1999	-0.806	-0.909	-0.548
Arable Land Rent 1999	-0.413	-1.038	0.211
% Green Party	-1.151	-0.101	-0.86
Share Grassland	-0.582	0.596	-1.463
Share Arable Land	0.107	-0.463	1.53
Share Agr. GDP	-2.303	1.273	0.823
Average Altitude	-0.339	0.959	-0.22
Share Rented Ag. Land	-0.625	1.462	-0.416
Pigs per ha	-1.485	-1.733	1.536
Cows per ha	-1.832	0.392	-0.901
Average Farmsize	-2.089	0.627	1.492

prices. Imbens and Hirano (2004) estimate a quadratic approximation including an intercept of the DRF based only on the GPS and the treatment by OLS. To account for further unobserved differences between the German states, we add state dummies. These differences could reflect macroeconomic conditions as well as local specificities from the implementation of agricultural policy. Morgan and Winship (2015) describe the approach of controlling for additional covariates after matching as “doubly robust”, although in the context of binary treatment variables. We estimate the equation

$$\ln R = f(N,\ GPS,\ S) \tag{3.6}$$

where R is district level rent, N is an indicator of Natura 2000 farming, GPS is the generalized propensity score, and S is a set of state dummies, by OLS.

3.5 Data

3.5.1 Data sources and variable choice

In 2010, according to the agricultural census (Farm Structure Survey – FSS), 59.8% of the utilized agricultural area (UAA) was rented. Therefore, we can assume that rental prices are a strong indicator for the value of agricultural land. Our analysis is cross-sectional in nature, but it uses results from two different agricultural censuses, namely 1999 and 2010. In particular, 1999 observations represent “pre-treatment” characteristics, i.e. district level agricultural characteristics before Natura 2000 farming policies were implemented (see also the discussion above). Similar to Michalek et al. (2014), we chose variables we believed affected the outcome (land rental price in 2010) as well as the treatment (Natura 2000 farming). Productivity is usually seen as a main driver of land prices, and therefore we control for 1999 district level productivity characteristics to model the generalized propensity score. Agricultural factors include the livestock densities of cows and pigs, as well as general land use variables such as arable land and grassland as a share of total agricultural land. Other productivity related characteristics may be captured by the pre-treatment (i.e. 1999) land rental prices. Farm size has also been an important driver of land rental prices by being able to exploit scale effects (Ciaian et al., 2012; Lence & Mishra, 2003; Michalek et al., 2014), as well as be an indicator for farmer lobbying power (larger farmers may also be better organized). The structure of the rental market, expressed as the share of rented land, has been argued to influence land sale prices by Feichtinger and Salhofer (2013), and the same argument could hold for rental prices as well. Finally, natural conditions such as the altitude above sea level are likely to affect land prices (higher altitudes are associated with rough terrain and less favorable climate conditions for many crops, thereby increasing production costs and decreasing productivity). Finally, in the FSS of 2010, Natura 2000 farming has been collected as an indicator variable equal to 1 if the farm received payments for Natura 2000 farming and 0 otherwise.

It is important to note that some of the sampling definitions changed between the FSS of 1999 and 2010, therefore direct comparisons cannot be made between the two datasets (for example by using a difference in difference approach). The following data sources were accessed for the analysis:

- District level data on average, grassland and arable land rents were acquired from the FSS 2010 and 1999 were collected from the state statistical offices of Germany. The regional statistics database (www.regionalstatistik.de) provided data on the amount of arable land, grassland, total agricultural land, livestock, and the amount of rented land in a district.

- The number of farms receiving Natura 2000 payments corresponding to the FSS 2010 was provided by the state statistical bureaus. On average, 6% of farms received payments for farming on Natura 2000 sites.

- Environmental protection policies are often associated with NGO activities and the green party. We use the district level results of the green party in the last state-level election as an indicator of how environmental protection is perceived by the general public.

- Data of regional GDP were accessed through the federal accounts database of the German Statistical Agency (http://www.vgrdl.de/)

- A digital elevation model (DEM) of Germany in 200m resolution, a shape-file showing the German districts, and Corine Land Cover data for Germany were downloaded from the German Geodata Center (http://www.geodatenzentrum.de). From the DEM, average altitude was calculated using QGIS (QGIS Development Team, 2015) for each district.

The summary statistics of our data are shown in Table 3.3.

3.5.2 Treatment of district restructuring in the data

The German states of Saxony and Saxony-Anhalt underwent a restructuring of their districts between 1999 and 2010, and Mecklenburg-Vorpommern in 2011. To improve

Table 3.3: Descriptive statistics

Variable	Unit	Mean	S.D.	Min	Max
Year 1999					
Average rent	€/ha	190	96	27	528
Grassland rent	€/ha	123	61	19	315
Arable land rent	€/ha	203	97	28	517
Share agr. GDP	Ag. GDP/total GDP	0.018	0.017	0.0001	0.075
Number of farms	Farms/district	716	657	10	3252
Agricultural land	ha/district	42302	40376	121	268935
Grassland share	Grassland/ag. area	0.32	0.23	0	0.99
Arable land share	Arable land/ag. area	0.65	0.223	0	0.99
Cow density	Animals/ha	0.81	0.55	0	2,36
Pig density	Animals/ha	1.14	1.637	0	13.063
Year 2010					
Average rent	€/ha	227	105	46	612
Grassland rent	€/ha	143	68.3	38	393
Arable land rent	€/ha	252	110.6	52	637
Altitude	meters	194.6	190	0.7	806.51
Share agr. GDP	Ag. GDP/total GDP	0.01	0.01	0	0.06
Share N2000 farms	N2000 farms/all farms	0.06	0.06	0	0.38
Number of farms	Farms/district	744	695	4	3572
Agricultural land	ha/district	46545	38411	158	175902
Grassland share	Grassland/ag. area	0.3	0.22	0	0.99
Arable land share	Arable land/ag. area	0.66	0.21	0.002	0.976
Cow density	Animals/ha	0.65	0.52	0	2.42
Pig density	Animals/ha	1.14	1.95	0	16.736

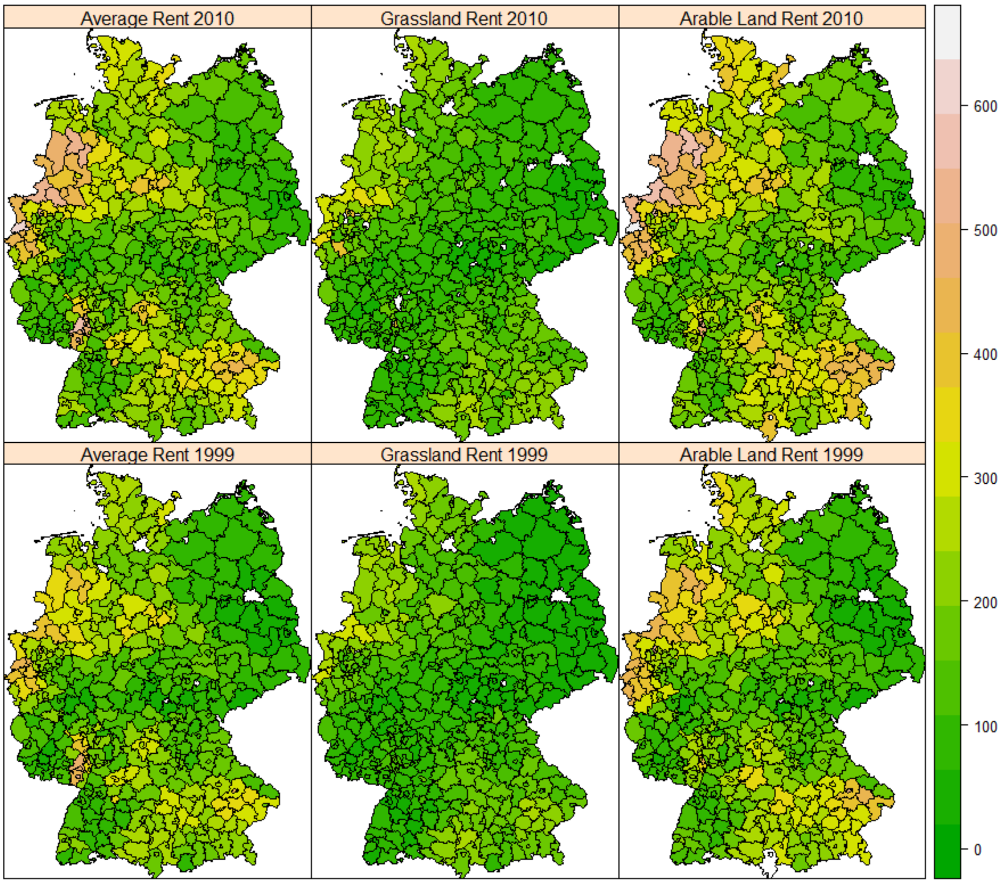

Figure 3.4: District average land rental prices in € per hectare in Germany in 2010 and 1999

the comparability of the two datasets, we applied the following procedure: Where two districts were merged, we took the average weighted by rented area (in case of rental price) or sum (e.g. hectares of farmland) of the two districts. If one district (a) was split into two and then merged with another district (b), we added a weighted average to of (a) to (b) and so on, based on the amount of land that was allocated to each district. We double-checked these results by comparing them to the 2010 data and found them to be similar.

3.6 Results and discussion

Agriculture in Germany is quite diverse, and so are rental prices as is shown in Figure 3.4. Part of the variation in rental prices can be explained by the division of Germany before 1989, and by differences in soil productivity. In particular, regions with high arable land rental prices in the North-West, as well as in the South-East, stick out. Relatively speaking, these patterns have remained largely unchanged as is clear from Figure 3.4.

If we compare Figure 3.4 to Figure 3.1, the latter of which shows the share of farms receiving payments for Natura 2000 farming, we see that the hot-spots of Natura 2000 farming are not in high-rental-price regions of Germany.

3.6.1 Econometric analysis

For each of our dependent variables (average rent, rent for grassland, rent for arable land) we estimate a separate Imbens and Hirano (2004) GPS model based on the zero inflated Beta (ZIB) distribution. As is described above, the model includes two stages. The first stage estimates the generalized propensity score (GPS), and the second stage estimates the outcome by OLS. The results of the zero inflated beta model for the GPS are presented in Table 3.4. Note that the ZIB model and therefore the estimated propensity score is identical in all three subsequent estimations of the outcome model.

We name the first part of the ZIB model the *Zero Model*, as it estimates the probability of having not a single Natura 2000 farm within a district. First, the rental prices in 1999 could be considered as an indicator of agricultural productivity. While the average rent is positively associated with a zero percentage of Natura 2000 farmers, grassland and arable land rents show a negative relationship. A higher percentage of voters associated with the green party, as well as higher share of grassland in 1999 decreases the probability of no Natura 2000 farming. Average altitude increases the probability of having no Natura 2000 farming.

The *Beta* model is interpreted conditional on the presence of Natura 2000 farming in a district. Here, the mean parameter is again affected by 1999 land rental prices, but here the signs are opposite to the Zero Model, and only significant for average rent and arable land rent. This means that given a positive percentage of Natura 2000 farmers, a higher average rent in 1999 will be associated with a lower percentage of Natura 2000 farmers in 2010, while a higher arable land rent is associated with a higher percentage of Natura 2000 farmers. Surprisingly, the Beta model suggests that a higher percentage of green party voters is associated with a lower percentage of Natura 2000 farmers. A possible explanation of this counter-intuitive result is that the presence/absence decision is made at higher policy level (i.e. state level), while the actual participation may still be protested by farmers. The green party is particularly strong in more urbanized districts, which may in turn have less potential for Natura 2000 site designation in general. Mainly rural districts, where voters are more generally conservative and vote for other parties than the greens, have more farmland that can be subject to Natura 2000.

A higher share of arable land is also associated with a lower percentage of Natura 2000 farmers, and so are the higher share of agricultural GDP and cow density. All three of these characteristics may be associated with highly productive agriculture, where either (1) natural conditions are not of community protection interest, or (2) farmers are more successful at lobbying against Natura 2000 designation. Interestingly, a higher share of pigs is associated with a higher percentage of Natura 2000 farmers. Finally, larger average farm sizes are also associated with more farmers being subject to Natura 2000 farming. Intuitively, one could assume that as a farm is larger, the chance of having some high-quality biodiversity land under cultivation is more likely as our theoretical model suggests.

In the second stage, we estimate the outcome model by regressing the log of rent on the GPS, the Natura 2000 indicator, and a set of state dummies. The results are shown in Table 3.5. The results clearly show a negative association between the Natura 2000 farming indicator and the log of land price for all three land categories. All models show heteroscedasticity robust standard errors computed using the sandwich estimator (Zeileis, 2004). More importantly, two out of three models also show that the terms involving the GPS are significant, which means that the covariates could

Table 3.4: Estimation results of the zero inflated Beta model used to compute the generalized propensity score. The dependent variable is the share of Natura 2000 farms in a district.

Model	*Beta model*						*Zero Model*		
Parameter (link)	Mean (logit)			Precision (log)			Alpha (logit)		
Variable	Estimate	Std. Err.		Estimate	Std. Err.		Estimate	Std. Err.	
(Intercept)	0.575	1.717		2.992	0.119	***	2.584	2.957	
Average Rent 1999	-0.009	0.005	*				0.047	0.013	***
Grassland Rent 1999	0.001	0.003					-0.014	0.007	*
Arable Land Rent 1999	0.007	0.004	*				-0.042	0.011	***
% Green Party	-4.73	2.28	**				-11.38	4.312	***
Share Grassland 1999	-0.992	1.641					-5.53	3.139	*
Share Arable Land 1999	-2.889	1.632	*				-1.884	2.734	
Share Agr. GDP 1999	-16.48	7.493	**				-4.996	17.58	
Average Altitude 1999	-0.001	0	***				0.004	0.001	***
Share Rented Ag. Land 1999	-0.338	0.488					0.516	1.243	
Pigs per ha 1999	0.187	0.066	***				-0.281	0.205	
Cows per ha 1999	-0.565	0.314	*				1.208	0.801	
Average Farmsize 1999	0.003	0.001	**				-0.001	0.003	
AIC					-355				
SBC					-258				
Observations					364				

Note: *p<0.1; **p<0.05; ***p<0.01

Table 3.5: Results of the outcome model regressing the log of 2010 rent on the Natura 2000 farm share and the generalized propensity score (GPS) by OLS using heteroscedasticity robust standard errors. Note that all models were estimated by including state dummies (not shown for brevity). Table generated with the `stargazer` `R` package (Hlavac, 2015)

	Dependent variable		
	Log Av. Rent	Log Grassland Rent	Log Arable Land Rent
Natura 2000 Farm Share	-2.546***	-1.652***	-2.018***
	(0.575)	(0.535)	(0.475)
GPS	-0.023**	-0.002	-0.022**
	(0.009)	(0.010)	(0.008)
GPS^2	0.001***	0.0001	0.001***
	(0.0002)	(0.0002)	(0.0002)
Natura 2000 Farm Share*GPS	-1.118	0.082	0.049
	(0.842)	(0.798)	(0.783)
Constant	4.842***	4.323***	4.754***
	(0.104)	(0.096)	(0.098)
Observations	265	265	265
R2	0.568	0.587	0.593
Adjusted R2	0.540	0.560	0.567
Residual Std. Error (df = 248)	0.330	0.292	0.312
F Statistic (df = 16; 248)	20.350***	21.993***	22.574***
Note:			*p<0.1; **p<0.05; ***p<0.01

indeed introduce some bias if not correctly handled by the matching procedure.

3.6.2 Impact of Natura 2000

The negativity of the Natura 2000 indicator suggests that indeed, Natura 2000 designation affects land prices negatively. Because the interaction between Natura 2000 farm share and the GPS is not significant in any of the models, we can interpret the parameters of the Natura 2000 estimator as the semi elasticity of rental prices with respect to Natura 2000. For example, a 1 percentage point increase in Natura 2000 farms will decrease average rental prices by 2.5%. This effect is relatively large, and in practice it may differ between the different Natura 2000 implementation models. Currently, the share of farmers receiving Natura 2000 payments is relatively low (6% on average), and our results should only be interpreted within the vicinity of current values. More responsive nonlinear functions could be estimated if more reliable data becomes available.

The impact on grassland rents is smaller than for average rents, and so is the effect for arable land rent. It suggests that rental prices of *other* land use types such as permanent crops could be particularly affected by Natura 2000 designation. We tested three indicators to describe the impact of Natura 2000 designation on land prices. For consistency, we used the Natura 2000 indicator derived from the farm structure survey rather than indicators constructed from Corine Land Cover (CLC) data. The CLC indicator may be imprecise, as data are generated from digitized large-scale aerial photographs and digitized to a 10 ha resolution.

The negativity of the total effect of Natura 2000 designation is consistent with the conjecture that the land designated to Natura 2000 should be used under protective and less intensive agricultural practice, which often shows relatively lower land productivity, in order to protect biodiversity. With regard to our theoretical model, the effect could be explained as follows. If the subsidy in company with potential input cost reductions does not sufficiently compensate the productivity loss, rents will be reduced. The effect will be stronger at the district aggregate level, if Natura 2000 designation does not increase competition for non-Natura 2000 farmland, which could push up average rental prices.

Our findings have implications for the future design of (agri-) environmental policy. As has been argued, farming and keeping open landscapes is seen as an integrative part of species conservation within Natura 2000 sites. But not fully compensated productivity impairments could lead to the abandonment of farming in marginal areas nonetheless, as farmers decide to stop cultivating their land. The lack of a profitable future of the business may increase difficulties in finding a successor (Bignal & McCracken, 2000; MacDonald et al., 2000; Visser, Moran, Regan, Gormally, & Skeffington, 2007). While the impact of farm abandonment on biodiversity is difficult to predict, the study by MacDonald et al. (2000) found that negative biodiversity impacts were to be expected in 15 out of 24 mountainous case study regions across Europe. In addition, abandonment of traditional farming practices may lead to monotonization or natural succession of landscapes. If the integrity of traditional landscape should be conserved in the long run (Plieninger, Höchtl, & Spek, 2006), strategies to preserve or improve traditional farming methods need to be developed. As Plieninger et al. (2006, p. 320) point out, "a sustainable landscape development is impossible without the involvement of land-users and local people, i.e. of the sculptors of the landscape".

Even though our results are in line with the theoretical arguments outlined in the literature, the magnitude of our findings cannot be directly compared to other studies. For example, while Letort and Temesgen (2014) also study the effect of environmental policy on land prices, the policy under investigation differs substantially from Natura 2000 designation in its focus as well as in breadth. In addition, we use a different method (the generalized propensity score), and we use district aggregate data rather than farm level data. Most studies on the incidence of subsidies on land values have applied some form of spatial regression model (e.g. Feichtinger & Salhofer, 2016; Letort & Temesgen, 2014), which can help to alleviate some spatial spillover effects (i.e. spatial lag and error terms), but not easily combined with other types of analysis (e.g. matching). While our results are robust and consistent with the theory, current measures in Natura 2000 farming are too diverse across states to provide more detailed policy recommendations from this aggregate study. Follow-up studies should examine specific programs at the farm level, possibly with data on actual farmer behavior rather than program prescriptions.

3.7 Concluding remarks

Protection of environmental resources such as biodiversity has become a major concern in the European Union. Agriculture can be a threat to biodiversity, but can also be used to foster it. In particular, traditional extensive farming methods can play a large role in protecting priority habitats and species (Gliessman, 2014; Ostermann, 1998).

However, protecting valuable farmland comes at a cost. Farmers are reduced in their capacity to make profit maximizing decisions and need to be compensated accordingly in order to keep farming marginal land. The literature has provided several theoretical and empirical explanations of how payments to farmers influence farmland prices (Ciaian et al., 2012; Ciaian et al., 2014; Feichtinger & Salhofer, 2016; Kilian et al., 2012; Michalek et al., 2014) and how environmental policy may influence land prices (Letort & Temesgen, 2014). We add to this growing body of knowledge by investigating the effect that designation of Natura 2000 protected areas and compensation payments has on farmland values as represented by their rental prices. By using generalized propensity score matching, we find a significant negative relationship between Natura 2000 farming and land rental prices.

Our results suggest that concerns of landowners and farmers were justified. Apart from increasing monetary incentives, authorities could support local producers in improving the marketability of Natura 2000 areas, e.g. through sustainable tourism (Hawkins, 2004; Mellon & Bramwell, 2016; Woodland & Acott, 2007), regional branding of products (Getzner, 2010; Hjalager & Johansen, 2013), or other strategies, if they are in line with biodiversity objectives. This could help to improve the acceptance of integrated conservation schemes such as Natura 2000. As we have seen from the literature, acceptance by the stakeholders (i.e. landowners and farmers) is a key aspect to effective conservation.

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Chapter 4

Highway Construction and Wildlife Populations: Evidence from Austria

Dieter Koemle, Yves Zinngrebe and Xiaohua Yu[1,2]

Abstract

Fragmentation and destruction of ecosystems due to highways is a key threat to habitat quality and biodiversity. In this article, we develop a theoretical framework and use a dynamic spatial panel data model to estimate how Austrian highway construction after 1968 has impacted the populations of roe deer, red deer and wild boar. The results indicate that a growing highway density leads to decreasing populations of roe deer and wild boar in their local district, contrasted with increasing populations in neighboring districts. Red deer populations were relatively insensitive to highway construction. Positive population effects in neighboring districts can be explained by the reduction of competition, disease transmission, and road kill. The results have important policy implications for Environmental Impact Assessments of infrastructure construction, particularly in the early stages of planning.

Key words: dynamic panel data, spatial lag model, ungulates, habitat fragmentation, habitat loss

4.1 Introduction

The construction of highways diminishes resources for many wildlife species globally (Fahrig & Rytwinski, 2009; Forman & Alexander, 1998; Newbold et al., 2015; van der Ree, Smith, & Grilo, 2015a; Völk & Glitzner, 2000; Völk & Wöss, 2001), and habitat fragmentation through linear infrastructure has been called the "single greatest threat to biodiversity" (Hess, 1996; Noss, 1991). In addition to the effects of habitat destruction, spillover effects from roads can reach far into the surrounding landscapes

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[2] The paper was jointly written by DK (80%) and YZ (20%). The idea was jointly developed by DK, YZ, and XY. Data were collected and analyzed by DK. XY provided comments on methodology.

(Baylis et al., 2016; Haddad et al., 2015). Similar to most developed countries, Austria has established a comprehensive highway system over the past 50 years. To make environmental impacts of these large infrastructure projects more transparent, many countries have adopted Environmental Impact Assessments (EIAs). Austria adopted EIA in 1993 (Umweltverträglichkeitsprüfungsgesetz 1993) and first applied it to highway construction in 1996. By 1996, a total of 1619 km of highway were built without being subject to EIA. However, particularly in highway development, the EIA has often been criticized of being of inadequate quality in order to prevent possibly detrimental effects on the environment (Duinker & Greig, 2006; Jaeger, 2015).

Highways impact wildlife populations mainly through two channels: habitat loss, which describes the reduction in quantity and quality of habitat, and habitat fragmentation, which describes the breaking apart of habitat. In addition, highway construction has also increased the exposure of wildlife species to wildlife-vehicle-collisions globally (Kušta, Keken, Ježek, Holá, & Šmíd, 2017). While habitat loss almost always has a negative effect on ecology, the evidence has shown that the impact of habitat fragmentation per se can be positive or negative (Fahrig, 2017). In this paper, we study the effects of highway construction on three ungulate species in Austria: roe deer *Capreolus capreolus*, red deer *Cervus elaphus*, and wild boar *Sus scrofa*.

Many factors influence the quality of a wildlife EIA in highway construction. First, there is uncertainty about the landscape scale effects and thresholds regarding infrastructure projects on wildlife, which often makes predictions difficult (Jaeger, 2015; Roedenbeck et al., 2007). These uncertainties are often not addressed in EIAs and therefore not incorporated into the decision processes. Second, wildlife species may be particularly sensitive to the cumulative impacts of a highway development project, which are often poorly addressed in EIA practice (Duinker & Greig, 2006; Masden, Fox, Furness, Bullman, & Haydon, 2010; Piper, 2001; Smith, 2006). Third, assessment procedures may not always strictly follow scientific standards, either due to political pressures, insufficiency of EIA guidance documents published by the relevant authorities, or lack of time and funding (Morrison-Saunders, Annandale, & Cappelluti, 2001)(Morrison-Saunders et al., 2001), as well as lack of competence and training of the personnel (Zhang, Kørnøv, & Christensen, 2013).

While the impact of highway development on wildlife has attracted plenty of research, the current literature mainly sheds light on the impact of infrastructure on wildlife habitat through specific channels and at small geographical scales. This is important from the perspective of ecological research. However, from a management perspective, the aggregate effect caused by a multiplicity of factors such as resource degradation, water and air pollution, noise, as well as impacts specific to the species under investigation such as changes in habitat size and fragmentation, and species interactions, is of more concern. This aggregate effect of road construction on animal populations has not been well studied at a national scale in a long time horizon, except for Roedenbeck and Köhler (2006), who studied the impact of landscape fragmentation on animal density in Hessen, Germany. In contrast, the present paper uses annual district level data from Austria after 1968 to evaluate the impacts of highway construction on the harvest densities of red deer, roe deer and wild boar, including neighborhood effects. In particular, we seek to (1) investigate methods and arguments used in highway construction EIAs in the context of wildlife in Austria, (2) propose a GIS-based method based on readily available data and an econometric framework to assess highway impacts on wildlife, that separates the dominating effects of habitat loss from fragmentation, and (3) discuss the results of the case study and give some recommendations for the future improvement of EIAs.

4.2 Background

In this paper, we investigate the ex post dynamic effect of highway construction on three ungulate species (roe deer, red deer, and wild boar) in Austria. These species have been subject to hunting for many decades, and therefore changes in populations will not only have ecological effects, but also economic effects as well.

4.2.1 Highway impacts on wildlife: ecological mechanisms from the literature

The impact of highways on wildlife has been studied comprehensively in the ecological literature, and a basic distinction is made between the effects of habitat loss and habitat fragmentation. The negative effect of habitat loss caused by highways can be explained by three mechanisms. First, constructing a highway causes direct habitat loss through sealing and hardening of surfaces and the removal of vegetation (van der Ree, Smith, & Grilo, 2015b). Second, highways through animal habitat increase light and noise pollution, air pollution through gas emissions (Huang, Bird, & Bell, 2009) and dust (Nanos & Ilias, 2007), and the runoff of salt and other chemical substances (Evink, 2002). Road avoidance as a behavioral response to noise and air pollution therefore may cause an additional loss of usable habitat (D'Amico, Périquet, Román, & Revilla, 2016; Laurian et al., 2008; Rost & Bailey, 1979). Depending on road width, traffic volume, the structure of the adjacent landscape, the nature of the prevailing wind, and the specific sensitivity of species to road effects, the road-effect zone (Forman, 1995) may extend far into the surrounding landscapes (Mäki, Kalliola, & Vuorinen, 2001; van der Ree et al., 2015b).

Third, as a further effect, highways may lead to increased development efforts in their vicinity (Selva, Switalski, Kreft, & Ibisch, 2015). In the literature, roads have been identified as being one important determinant of deforestation (Chomitz & Gray, 1999; Deng, Huang, Huang, Rozelle, & Gibson, 2011). Mothorpe, Hanson, and Schnier (2013) find that the construction of the interstate highway system in Georgia, U.S. has caused substantial losses in agricultural land due to residential development. For Austria, Figure 4.1 indicates a similar relationship by showing a positive relationship between the density of highways (km/km^2) and human population density.

Classical ecology assumes that fragmentation reduces an animal's potential to move freely according to the availability of the fundamental resources food, water, and shelter (Benz et al., 2016; Morrison, Marcot, & Mannan, 2012). Several studies have tried to link population decline to habitat fragmentation, e.g. for European hare in Switzerland, Austria, and Czech Republic (EEA, 2011), or badgers in the Netherlands (Fahrig, 2002). In Germany, roe deer densities were positively correlated with effective mesh size (Jaeger, 2015), indicating that less fragmented landscapes support larger roe deer populations (Roedenbeck & Köhler, 2006).

Contrasting these negative effects, a review article by Fahrig (2017) finds that 76% of 381 significant ecological responses to habitat fragmentation per se in 118 case studies were positive. Hess (1996) argues that fragmentation might stop the

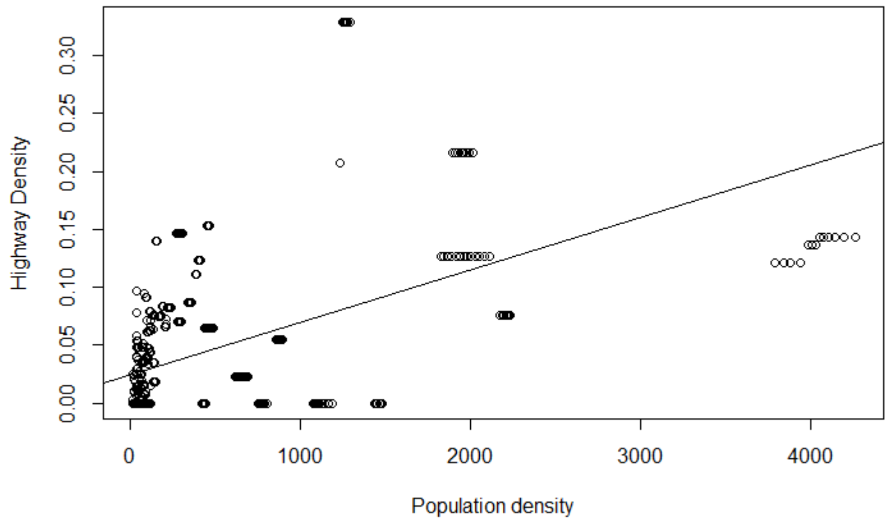

Figure 4.1: Density of highways plotted against human population density (years 2002-2014; Source: Statistik Austria)

transmission of contagious diseases among animal populations. Studies on infectious diseases in wildlife in Austria suggest that swine fever and brucellosis are a problem in wild boar (Reimoser & Reimoser, 2010), while paratuberculosis has been found in red deer (Fink et al., 2015; Schoepf et al., 2012) and roe deer (see Duscher, Leschnik, Fuehrer, and Joachim (2015) for a recent review of the literature).

Additionally, highways in Austria are fenced, so that road kills on highways are practically negligible compared to those on rural, lower-order roads. As highways also offer more convenient ways of transportation than lower-order roads, a diversion of traffic may reduce road kill. Kušta et al. (2017) find that ungulate-vehicle collisions are most frequent on first-class, second class roads compared to motorways and expressways in Czech Republic. Figure 4.2 shows that road kills in Austria decrease with a higher highway density for roe deer and red deer, but increase for wild boar. Given regular fence maintenance, fencing may be particularly beneficial for population persistence when road avoidance of a species is low and traffic mortality is high (Jaeger & Fahrig, 2004).

Finally, separating two habitats by a highway may decrease the intra-and interspecific competition effect that a species experiences (Fahrig, 2017). Separation of habitats could lead to a sudden decrease in interference competition (Begon, Townsend, & Harper, 2005), which could in turn increase population densities. Predator-prey dynamics will change if the predator is more negatively affected by a road than the prey species. In this case, there may be a positive abundance effect for the prey species (Fahrig & Rytwinski, 2009; Liao, Bearup, & Blasius, 2017).

Nevertheless, whether or not a species responds to highway construction depends on home range size, habitat characteristics (vegetation, geology and climate), movement patterns (e.g. seasonal migration) and other (e.g. human) interference (e.g. feeding or hunting), as well as the ability to adapt to new conditions.

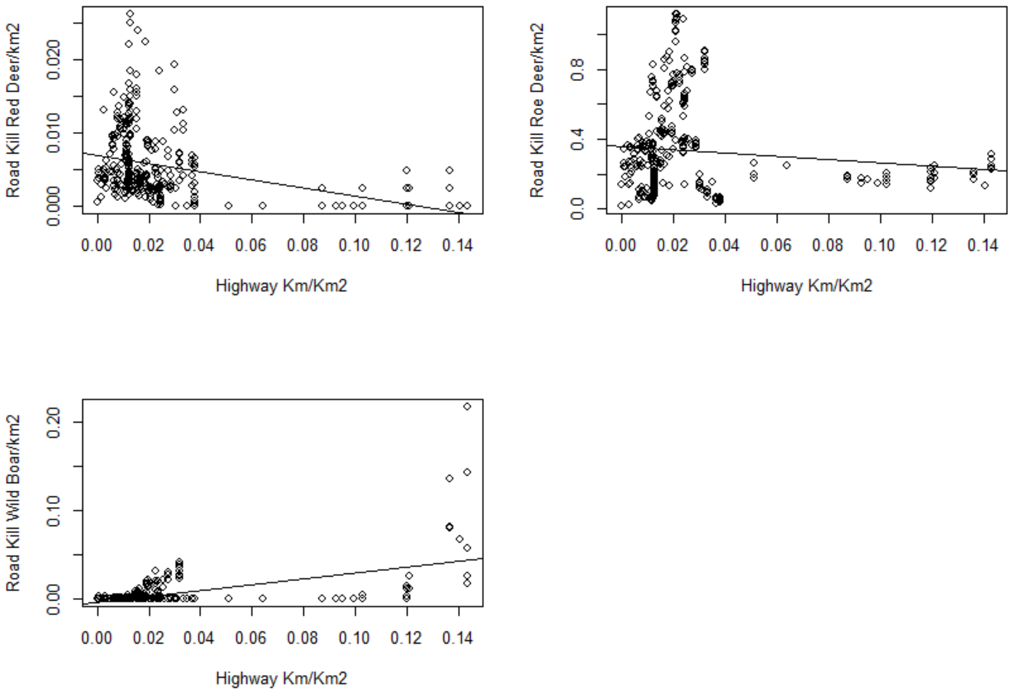

Figure 4.2: : Road kill per km^2 of red deer, roe deer and wild boar plotted against the density of fenced roads in Austria (province level 1968-2014; Source: Statistik Austria)

4.2.2 Highway construction in Austria

Austria is a country in the center of Europe with around 8.5 million inhabitants and a total land area of about 84 thousand km^2. As of 2012, Austria is separated into 95 districts. We use the term highway for both top order road types, "Autobahnen" and "Schnellstraßen", which are similar in width, construction, fencing, and speed limits, and therefore likely to have similar effects on wildlife populations. The first sections of highway were built during the Nazi regime along the Salzburg – Linz – Vienna connection (today highway A1 "Westautobahn"). Building activities were suspended by the end of 1941 with only 16.8 km finished close to Salzburg. Highway construction was continued from 1954. The Austrian Federal Road Act of 1971 (Bundesstraßengesetz) marked the peak of highway planning activities, leading to a planned total of 1874 km of highways on the Austrian territory (ASFINAG, 2012a).

The first critical voices about highway construction were echoed during the early 1980s. The rising ecological movement, as well as funding problems, led to open protests against motorways that were currently in the planning or construction stage. According to a report by the Austrian road construction and financing authority (ASFINAG), new highways were reassessed and environmentally less harmful features (tunnels) and highway overpasses to compensate for their ecological impacts were introduced. The trend is shown in Figure 4.3, where up until around 1990, a sharp rise in the density of highways can be observed, with a leveling off of new highway openings after 1995 in most provinces.

Today Austria is an important transit country between western and eastern Europe, as well as from north to south (Zink and Reimoser, 2008). This includes four corridors

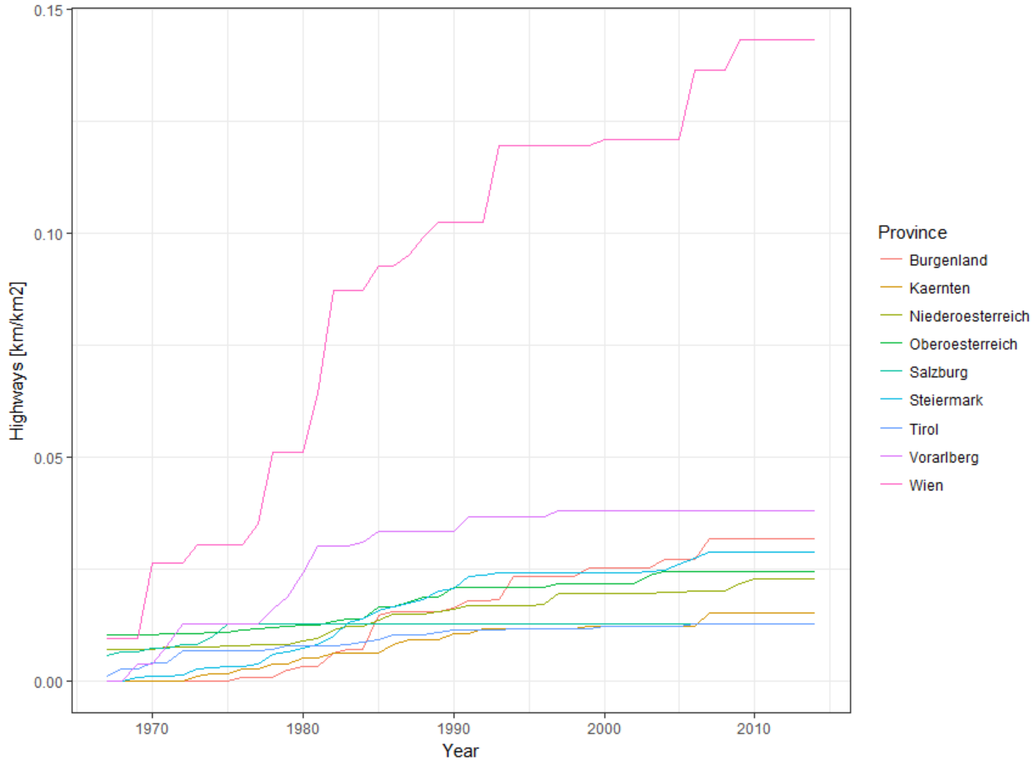

Figure 4.3: Changes in cumulative highway density in the nine provinces of Austria (Source: ASFINAG 2012, own calculations)

of the core Trans-European Network Transport (TEN-T), with a total length of 1072 km: Baltic-Adriatic, Rhine-Danube, Scandinavia-Mediterranean, and Orient/East-Mediterranean. The total comprehensive TEN-T in Austria is 1689 km (CEDR, 2016). Therefore, the construction of suitable highways is a priority not only nationally, but also at the European level. 2185 km of highways are in operation, 26 km are currently in construction, and 31 km are planned (bmvit, 2016). Figure 4.4 shows the current spatial distribution of highways in Austria.

4.2.3 Ungulates and Habitat Connectivity in Austria

Among the native ungulate species in Austria, roe deer, red deer, and wild boar are among the most important ones in terms of harvest numbers. Their average harvest density for 1968-2014 for Austrian districts is shown in Figure 4.5.

Red deer are mostly found in large, connected nemoral deciduous forests. However, some recent populations have also survived in rather small, local wooded areas in Austria (Bauer, 2001b). They are described as intermediate feeders (Hofmann, 1989) ingesting a mixture of concentrate foods and crude fibers. Because red deer require substantial amounts of food to meet their physiological needs, they are often required to migrate over large distances. In contrast, if food is abundant, migration may not be necessary (Keken & Kušta, 2017; Kušta et al., 2017). According to Bauer (2001b) and Schmidt (2014), densities have increased in certain areas because of intense winter feeding

Roe deer is very adaptable to many types of habitat, and it has made habitat from natural small-structured diverse forests, to intensively used agricultural landscapes

Figure 4.4: Current (2016) highway system in Austria (Source: OpenStreetMap)

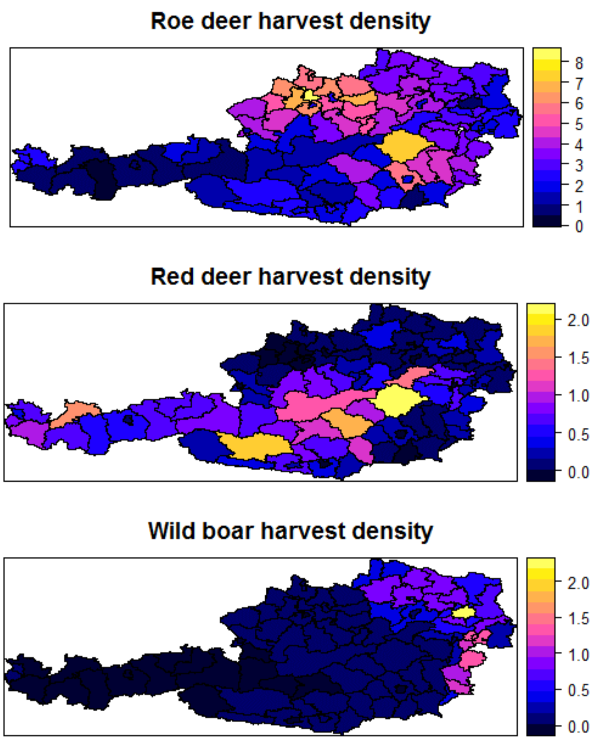

Figure 4.5: Distribution of the average 1968-2014 harvest density (individuals/km2) of red deer, roe deer, and wild boar in Austria (Source: Statistik Austria)

its home (Bauer, 2001a). Roe deer is generally described as a concentrate selector (Hofmann, 1989). In Austria, roe deer is found in all districts, but it reaches its highest densities in the agriculturally dominated east and north (Figure 4.5).

Wild boar today is mostly present in the eastern parts of Austria (Figure 4.5). Throughout history, it has often been subject to management prescriptions due to damages it caused in agriculture and forests (Bauer, 2001c). It has shown a substantial increase in population size over the past two decades, particularly in the provinces of Lower Austria and Burgenland, and our harvest data show that wild boar has spread westward towards the alps.

Völk and Glitzner (2000) use red deer, roe deer, and wild boar and other indicator species to study how well crossing structures for wildlife work. Their findings indicate that highways have a particularly severe barrier effect in flat, intensively used agricultural areas with low forest cover. On the other hand, barrier effects in mountainous areas are mostly due to poor design of wildlife passages. Several authors have criticized that planning agencies often have not adequately considered landscape-scale functional relationships when developing wildlife crossings (Keken & Kušta, 2017; Völk & Glitzner, 2000). The focus of minimizing noise disturbance for humans when planning new roads makes it more likely that high-traffic roads are planned in environmentally sensitive areas. Based on footprint analyses and interviews with local hunters, Völk and Glitzner (2000) report that roe deer frequently used all available crossing structures, while other species (e.g. wild boar or red deer) were more selective with their choice of crossing structures.

4.2.4 Wildlife assessments in Austrian EIAs

EIAs were first introduced by the US Environmental agency NEPA (National Environmental Protection Act 1970 in the United States). Ever since, countries develop projects that require an EIA to assess and compare environmental impacts of road construction, mining projects, or other major construction endeavors. In general, an EIA identifies and assesses impacts, compares them to alternative scenarios for how to develop a project and then informs the decision on the project approval. A good EIA requires a team of experts, sufficient amounts of data, and capacities for data analyses to provide sufficient evidence for ex-ante project evaluation and impact prevention and minimization. After implementation, systematic monitoring and evaluation needs to be carried out in order to validate assumptions made in the ex-ante assessment, and to be able to compensate for any additional negative impacts (Glasson, Therivel, & Chadwick, 2013).

In Austria, EIAs have been mandatory after 1993, and the first highway project subject to EIA was started in 1996 . While assessing all relevant EIAs would go beyond the scope of this paper, we give a short overview over the six most recent highway project EIAs. The way wildlife impacts are assessed in those EIAs can be distinguished in three dimensions:

(i) in the way they frame and value impacts on wildlife;

(ii) in the way those impacts are quantified and assessed and

(iii) the recommendations and follow-ups that result from the assessments.

The impact on hunting conditions is a central aspect assessed by EIAs. The EIA for "Mühlviertler Schnellstraße" evaluates "hunting attractiveness" and sensitivity of impact along the criteria: spectra of wildlife species, habitat conditions, barriers to wildlife crossing, damage caused by wildlife, and hunting attractiveness (REVITAL ecoconsult, 2007). Additionally, population and habitat loss are evaluated. Fürstenfelder Schnellstraße EIA (Depisch, Raderbauer, Grulich, Schmetta, & Paill, 2008) similarly assesses wildlife spectrum, habitats, wildlife passes (regional and local) and barriers, wildlife damage reports, and hunting methods and the attractiveness of hunting (e.g. annual drive-hunts etc.). Similar criteria with a focus on hunting quality were equally listed in the S1 EIS (Barbl, 2009). In the case of the A5 Nordautobahn EIA (Barbl, 2005), impacts of light, noise, vibration, dust and area reduction on wildlife were estimated as low without specifying methodological approaches or scientific references.

Assessment methods applied include local mapping of habitat characteristics and hunting equipment (e.g. feeding stations or hides), compiling statistics on harvest, road kill and wildlife diseases, accessing EIA legal documents and government reports, and to a smaller extent also reviewing scientific and grey literature. Only a small amount of effort was put into the study and prediction of behavioral responses. All EIAs heavily relied on interviews with local hunters, and some also note public participation processes that would allow residents to voice their concerns. Predictions in EIAs were separated for the construction phase and the operation phase. Most predictions, e.g. with respect to wildlife behavior, were based on qualitative assessments.

All six EIAs we studied concluded, that the highway project only had minor effects on hunting and game species. Some impacts were "downgraded" after considering compensation measures, such as the barrier effect of S3 Weinviertler Schnellstraße (ASFINAG, 2012b), which was reduced to from "high" to "medium" due to proposed crossing structures.

Despite all the EIAs announcing the need for follow-up procedures and monitoring, no systematic process of supervising and assessing the implementation of suggested follow-up has been documented in either of the processes. This adds to findings in literature observing i.a. a lack of guidance, baseline data and defined monitoring procedures (Arts & Nooteboom, 1999).

Jaeger (2015, p. 35) summarizes the lack of quality in EIAs as "(i) most EIAs are too vague or make unsubstantiated predictions, (ii) most EIAs do not consider the landscape scale, and (iii) almost none use state-of-the art modeling methods to predict likely effects". Furthermore, EIAs have been criticized for not applying scientifically consolidated and politically legitimized values and measures and have instead been observed to individually, sometimes randomly define values and evaluation criteria (Beattie, 1995). While the literature emphasizes the need to assure participation of all stakeholders (Glasson et al., 2013, e.g.), Austrian procedures reveal a strong bias towards certain interest groups, such as hunters. Besides the need for a wider and more representative stakeholder involvement, the development of standardized measures and reference base line data can help increasing the transparency and legitimacy of EIA procedures.

4.3 Data and methods

4.3.1 Theoretical framework

The well-known Schaefer model describes population dynamics as a logistic growth model minus annual harvest (Conrad & Clark, 1987; Schaefer, 1957)

$$\frac{dN_t}{dt} = r_t N_t \left(1 - \frac{N_t}{K_t}\right) - H_t \tag{4.1}$$

which describes the dynamics of population abundance N_t over time t. The parameters governing the population dynamics in the logistic growth model are the intrinsic growth rate r_t and the environmental carrying capacity K_t (Pastor, 2009). Shifts in r_t and K_t change population dynamics and equilibria. Annual harvest H_t is described by

$$H_t = qN_t E_t \tag{4.2}$$

where q is a catchability coefficient and E_t is hunting effort. In equilibrium, where $\frac{dN_t}{dt} = 0$, harvest equals annual growth, such that $r_t N_t^* \left(1 - \frac{N_t^*}{K_t}\right) = qN_t^* E_t$ which can be solved for N_t and then substituted back into (4.2)

$$H_t^* = qKE_t - \frac{q^2 K_t}{r_t} E^2 = \left(q - \frac{q^2 E_t}{r_t} \right) K_t \ E_t \tag{4.3}$$

or more generally, the optimal harvest is

$$H_t^* = H\left(K_t, r_t, E_t\right). \tag{4.4}$$

Clearly, carrying capacity K_t and intrinsic growth rate r_t are vulnerable to the environment, and road construction will inevitably affect these two variables. Particularly, due to high mobility of animals and regional resource competition, both local highway construction and highway construction in neighbor regions could impact these variables. Taking into account highway construction, Equation (4.4) could be rewritten as

$$H^* = f\left(r\left(D, ND\right), K\left(D,\ ND\right),\ E\right) \tag{4.5}$$

The marginal effect of highways on equilibrium harvest can therefore be separated into

$$\frac{dH^*}{dD} = \frac{\partial f}{\partial r}\frac{\partial r}{\partial D} + \frac{\partial f}{\partial K}\frac{\partial K}{\partial D} \tag{4.6}$$

and

$$\frac{dH^*}{dND} = \frac{\partial f}{\partial r}\frac{\partial r}{\partial ND} + \frac{\partial f}{\partial K}\frac{\partial K}{\partial ND} \tag{4.7}$$

Where D is local highway density and ND is neighbor district highway density. $\frac{dH^*}{dD}$ is the total within-district effect of a change in highway density on equilibrium harvest.

$\frac{\partial f}{\partial r}\frac{\partial r}{\partial D}$ is the change in harvest caused by the effect of highway density on r. Habitat loss within a district will likely make $\frac{\partial r}{\partial D} < 0$, and $\frac{\partial f}{\partial r} > 0$, therefore this effect will be negative.

$\frac{\partial f}{\partial K}\frac{\partial K}{\partial D}$ is the change in harvest caused by the effect of highway density via K. Again, this effect is likely to be negative, because $\frac{\partial K}{\partial D} < 0$ and $\frac{\partial f}{\partial K} > 0$.

To separate the effects of fragmentation from habitat loss, we also include the effect of neighbor district highway density on equilibrium harvest.

$\frac{dH^*}{dND}$ is the total effect of a change in highway density on harvest density in its neighbor districts.

$\frac{\partial f}{\partial r}\frac{\partial r}{\partial ND}$ is the effect of a change in highway density in a neighbor district on harvest via the intrinsic growth rate r, where $\frac{dr}{dND}$ could be positive or negative. $\frac{dr}{dND}$ will be positive if effects that increase the growth rate (e.g. the diversion of traffic towards wildlife-proof roads or a lower chance of spreading infectious diseases) outweigh the negative effects of fragmentation (e.g. difficulty in finding a mate in the separated landscape).

$\frac{\partial f}{\partial K}\frac{dK}{dND}$ is the effect of a change in highway density in a neighbor district on harvest via carrying capacity, where $\frac{dK}{dND}$ could be positive or negative. Carrying capacity could increase e.g. if development is diverted towards districts with higher highway densities and abandoned in neighbor districts.

The total effect on neighbor districts could therefore be positive if the benefits from fragmentation outweigh its negative effects.

4.3.2 Variable selection and estimation strategy

As a dependent variable, we use the annual harvest data at the district level for roe deer, red deer, and wild boar. Our independent variable of interest is the cumulative highway density. Both data series are available throughout the years 1968-2014. Further, it is important to control for other factors that influence the equilibrium population levels. The availability of resources (water, food, shelter) depends on abiotic factors such as precipitation and temperature, and biotic factors such as competition from other species that occupy similar ecological niches (Birch, 1957; Putman, 1996; Richard, Gaillard, Saïd, Hamann, & Klein, 2009). The panel data structure allows us to include district-specific fixed effects related to time-invariant habitat conditions. This includes geological features (mountains, valleys) which can be seen as a proxy for possible crossing structures. For example, a mountainous region is likely to have more bridges and tunnels where wildlife species can cross.

As the Schaefer model suggests, human behavior can also influence wildlife species. This may include the amount of effort put into harvesting, which we approximate by the number of hunting licenses each district.

Because the effects of hunting are dynamic both in space and in scale, we include spatial and spatial and temporal lags of annual harvest into our model. Hunting regulations in Austria are governed by the nine Austrian provinces. Differences in regulations include for example timing and length of hunting seasons. These effects are removed when first differences are constructed before the estimation.

Finally, we added a dummy equal to one if the observation was made after 1995 and zero otherwise. Interacting this dummy with the highway density variable allows us to see whether there is a significant change in the effect of highway construction with the mandatory EIAs in place.

From the arguments outlined in above, we specify our estimation equation as

$$\begin{aligned}
H_{i,t} =& c_t + f_i + \gamma_1 Temperature_{i,t} + \gamma_3 Precipitation_{i,t} + \sum_k \gamma_{3,k} Competitor_{k,i,t} \\
&+ \gamma_{41} D_{i,t} + \gamma_{51} \left[WND_{i,t}\right] + \gamma_{42} D_{i,t}\ I95_{i,t} + \gamma_{52} \left[WND_{i,t}\right]\ I95_{i,t} \\
&+ \gamma_6 H_{i,t-1} + \gamma_7 \left[WH_{i,t}\right] + \gamma_8 licenses_{i,t}
\end{aligned} \tag{4.8}$$

where $H_{i,t}$ is harvest density in district i at time t, c_t is a time-fixed effect, $Competitor_{k,i,t}$ is the harvest density of k competing species, D is within-district highway density, ND is the neighbor district highway density, W is a spatial weights matrix, and *licenses* is the number of hunting licenses per square kilometer. $I95_{i,t}$ is an indicator variable that equals *one* if the year is after 1995, and *zero* otherwise. This model specification can be described as a Spatial Lag Model (LeSage and Pace, 2009) combined with an AR(1) panel data model. Our specification includes several issues that make standard OLS techniques infeasible. To account for intertemporal harvest dynamics, we included a lagged dependent variable in our model specification ($H_{i,t-1}$). This own-district lagged dependent variable creates an endogeneity problem that forbids estimating the model with a standard spatial random effects or fixed effects model. An efficient estimator is the well-known Arellano-Bond estimator (Arellano & Bond, 1991), which uses first differences to remove the individual-specific effect f_i and uses higher-order lagged dependent variables as instruments to remove the endogeneity problem (see Baltagi, 2005, p. 149f).

In addition, spatial panel data models integrate spatial information, in particular neighbor relations, into panel data models (Baltagi, 2005). To account for spatial relationships, the researcher needs to choose a spatial weights matrix. In principle, spatial weights matrices based on distance and on contiguity can be distinguished. Because of our data structure, we decided that the spatial contiguity matrix would be the most appropriate in our application. The spatial weights matrix was generated based on a district-level shape file of Austria, using the R package `spdep` (Bivand, Pebesma, & Gómez-Rubio, 2013; Bivand & Piras, 2015). Only first-order "queen" neighbors were used. The spatial lag was then computed as the average of all neighbor variables, e.g. the spatial lag of highway density is the average of the highway densities in all surrounding districts. Data preparation, such as merging different datasets, was conducted in `R` (R Core Team, 2014). To account for spatial relationships of species, we include the spatial lag of the dependent variable. For consistency with our theoretical model, we include the spatial lag of highway density. We interpret this spatial lag as an indicator of the fragmentation effect as opposed to the habitat loss effect.

After merging all data, we used STATA version 13 (StataCorp, 2013) to estimate our model using the `xtabond` function. The validity of our models' instruments was tested using the Sargan test for overidentification, and the Arellano-Bond test for autocorrelated residuals.

4.3.3 Data

The data for estimation Equation 4.8 were collected from several sources. Descriptive statistics are presented in Table 4.

- The statistical agency of Austria provides district-level hunting records dating back to the late 1940s. However, the re-structuring of districts in the years following World War II, led us to discard the first 20 years and start our analysis with the year 1968. From this year onward, we are also able to use the number of hunting licenses issued in a specific province.

- Geographical data were accessed through the OpenData portal of Austria (https://www.data.gv.at/), where we used a shapefile showing municipality-level borders. Municipality polygons were combined into district polygons using the R package `maptools` (Bivand & Lewin-Koh, 2016). From this, we generated the spatial weights matrix used in the subsequent analysis.

- Data on annual highway density were generated from a report by the Austrian road construction and financing authority (ASFINAG, 2012a). Each individual highway section was extracted and its length measured from Google Maps (http://maps.google.com). Highway sections were added to the dataset for all years after their official opening to public use. The kilometers of highway were then accumulated by year, and normalized by the total area of the district to establish the highway density (km/km2) indicator (Forman & Alexander, 1998).

- Geo-referenced climate variables for 1968-2014 were accessed through the website of the HistAlp project (www.zamg.ac.at/histalp) (Auer et al., 2007). This dataset includes geo-referenced monthly temperature and precipitation measurements from 61 meteorological stations distributed all over Austria. Annual averages for each district were calculated in `R` (R Core Team, 2014) using the following procedure. A 1000*1000 field grid was overlayed over a shapefile of Austria. Next, by using inverse distance weighting (Pebesma, 2004) the measured temperature and precipitation values were interpolated and projected onto this grid. Finally, for each district and year, an average was calculated from the interpolated values.

Table 4.1: Descriptive statistics

Variable	Unit	# Obs	Mean	Std. Dev.	Min.	Max.
Roe deer density	harvest/km2	4560	2.98	1.91	0	12.89
Red deer density	harvest/km2	4560	0.37	0.49	0	4.88
Wild boar density	harvest/km2	4560	0.24	0.57	0	5.47
Highway density	km/km2	4560	0.02	0.04	0	0.33
Average Temp.	1/10 °C	4560	76.84	15.62	24.49	117.63
Annual Precip.	mm	4560	925.68	215.35	320.63	1894.52
Hunting licenses	licenses/km2	4560	1.3	0.33	0.08	3.26

4.4 Results and discussion

4.4.1 Results

The results of three Arellano-Bond models are shown in Table 4.2. Different model specifications were tested to find out how sensitive parameters reacted. The models were then chosen based on the results of the Sargan test. In the following, the terms positive and negative are strictly related to the signs of the estimated parameters and do not infer any value judgements.

All three species are positively affected by their own temporal lags, as well as their own spatial lags. Also, roe deer harvest density is positively correlated with the contemporaneous harvest density of wild boar. Red deer, on the other hand, is positively correlated with the contemporaneous harvest density of roe deer. Finally, wild boar is contemporaneously correlated with the harvest density of roe deer.

Regarding our research questions, we find consistent results. All three species react to highway construction, but in different ways. For roe deer, the own-district effect of highways is negative (-1.42), and becomes even more negative after Austria's accession to the EU in 1995 (-1.974). On the other hand, the neighbor district effect of highways is positive for roe deer (4.434), and becomes even larger after 1995 (3.616).

Red deer is the most insensitive to highway density according to our results; only the own-district effect of highway density after 1995 was significantly negative (-0.22).

Wild boar is also negatively affected by highway in the same district (-1.113), but positively affected by highways in neighbor districts (3.339). However, wild boar does not show any significant reactions to the introduction of EIA in highway construction after 1995.

4.4.2 Explaining the observed relationships

The effects of habitat loss are negative, even at a district scale, as has been described by Fahrig (2017) and many others. For roe deer and wild boar, if positive fragmentation effects on population abundance exist at the district level, they may be overwhelmed by the effects of habitat destruction, noise, emissions, and contagious development. Interestingly, habitat loss effects after 1995 have not been compensated despite the requirement to assess the environmental impacts of highways. On the contrary, the effect on red deer has even gone from neutral to negative after 1995.

The positive effect of neighbor highway density on harvest of roe deer and wild boar, could be explained by a dominance of positive fragmentation effects with increasing distance from a highway. All the effects outlined above, such as (1) reduced spread of infectious diseases (Hess, 1996), (2) reduced competition (Fahrig, 2017), or (3) the reduction in road kill could explain this effect. Highways increase the costs of moving through the landscape, because animals need to search for an over-or underpass. Regarding (1), hunters could benefit from higher densities by achieving higher hunting successes, and also harvest qualitatively higher meat if the spread of infectious diseases is contained. (2) implies that higher densities can be achieved if less animals are able to interact in total due to the barrier. Interactions with other individuals from the same species, including fights over females and for forage, require energy and could reduce winter survival probabilities. In addition, inter-specific competition and predation could be reduced, as wild boar have been found to prey on roe deer fawns. (3) implies that the environment becomes less risky for ungulates with regard

Table 4.2: Results of Arellano-Bond models estimating the influence of highway density on harvest density of roe deer, red deer, and wild boar. Models (1) and (2) are one-step models, while model (3) was estimated using the two-step method. Model selection was based on the Sargan test. See Table 1 for variable descriptions. “Neighbor” describes the spatial lag, while “Lag” describes the temporal lag.

	Dependent variable		
	Roe deer density	Red deer density	Wild boar density
	(1)	(2)	(3)
Highway Variables			
Highway density	-1.420**	-0.0705	-1.113**
	(0.674)	(0.192)	(0.486)
Highways after 1995	-1.974***	-0.220*	-0.283
	(0.410)	(0.116)	(0.374)
Neighbor Highways	4.434**	0.0697	3.339***
	(1.751)	(0.501)	(0.494)
Neighbor Hw. after 1995	3.616***	0.0391	0.0263
	(0.820)	(0.235)	(0.419)
Species effects			
Roe deer density		0.00483	0.00711***
		(0.00302)	(0.000485)
Lag roe deer density	0.529***		
	(0.0123)		
Neighbor roe deer dens.	0.436***		
	(0.0161)		
Red deer density	0.121***		-0.00356
	(0.0412)		(0.00458)
Lag red deer density		0.689***	
		(0.0117)	
Neighbor red deer dens.		0.205***	
		(0.0168)	
Wild boar density	0.0255	0.0129**	
	(0.0217)	(0.00615)	
Lag wild boar density			0.322***
			(0.00311)
Neighbor wild boar dens.			0.781***
			(0.00550)
Environmental variables			
Temperature	0.000873	-0.000507	5.35e-05***
	(0.00122)	(0.000349)	(1.12e-05)
Precipitation	-2.78e-05	3.36e-05**	3.08e-05***
	(6.01e-05)	(1.71e-05)	(1.34e-06)
Hunting Licenses	0.0846	0.00843	0.0454***
	(0.0623)	(0.0178)	(0.00327)

	Dependent variable		
	Roe deer density	Red deer density	Wild boar density
Decade dummies (base years: 1968-1969)			
Year 1970-1979	-0.0236	0.0126	-0.0357***
	(0.0387)	(0.0110)	(0.00960)
Year 1980-1989	-0.104**	-0.00194	-0.0613***
	(0.0464)	(0.0132)	(0.00930)
Year 1990-1999	-0.113**	-0.00347	-0.0839***
	(0.0531)	(0.0150)	(0.00947)
Year 2000-2009	-0.154***	0.00869	-0.127***
	(0.0591)	(0.0166)	(0.00951)
Year 2010-2014	-0.167***	0.00871	-0.134***
	(0.0643)	(0.0183)	(0.00930)
Constant	-0.158	0.00795	-0.0959***
	(0.119)	(0.0338)	(0.00932)
Observations	4,370	4,370	4,370
Number of districts	95	95	95
Sargan Test Chi^2	2836.85	2974.38	84.58
(p-value)	(0.8243)	(0.1914)	(1.000)
1^{st} order Arellano-Bond Test			-3.2534
(p-value)			(0.0011)
2^{nd} order Arellano-Bond Test			.99273
(p-value)			(0.3208)
Note:			*p<0.1; **p<0.05; ***p<0.01

to vehicle collisions. This, however, should only be interpreted in the given context. The fragments created by highways are still relatively large, and extending fencing to lower order roads could exacerbate the negative fragmentation effects, as animals are not able to find enough resources in smaller and smaller fragments.

Another study that used road density as an indicator for fragmentation showed no significant impact of fragmentation per se on lynx populations in Canada (Hornseth et al., 2014), while Roedenbeck and Köhler (2006) found significant negative impacts of effective mesh size on roe deer, red deer, wild boar, and fox harvest densities in Germany. However, while Hornseth et al. (2014) controlled for habitat loss, Roedenbeck and Köhler (2006) did not. No correlation between neighbor highway density and red deer populations suggests that red deer is not strongly affected by fragmentation. There are however alternative explanations. One possible explanation could be that within a district, red deer mostly occurs in areas that are unsuitable for highway construction. In addition, feeding practices may have decoupled population persistence from migratory behavior (Schmidt, 2014). The additional development after 1995 may have affected red deer habitat. To reduce the impact of roads on wildlife in the future, some authors have advocated for the maintenance of road-free areas to reduce contagious development and other effects (Selva et al., 2015) by considering road-free areas in planning processes.

While it is beyond the scope of this study to explain the detailed ecological mechanisms that lead to these results, we can draw some general implications from our findings. First of all, increases in ungulate populations can be associated with economic benefits related to an increase in hunting opportunities, therefore providing income in rural areas with lower highway access. However, economic losses could ensue from an overpopulation of some ungulate species, for example in forestry through bark-stripping by red deer or damages in agriculture by wild boar. In addition, increased densities of roe deer can affect the diversity of native plants and herbs, leading to possible cascading effects within the ecosystem (Côté, Rooney, Tremblay, Dussault, & Waller, 2004; Jirinec, Cristol, & Leu, 2017).

Obviously, our model has some limitations. The first comes from the available data. While it would be important to include land cover data such as forests or arable land to account for other aspects of habitat quality, these were not available in a consistent form over the full time frame. Agricultural practices are particularly important, as the consolidation of farms and agricultural land, as well as the changes in crops grown and the use of agrochemicals have modified habitats considerably. However, systematic land use assessments, such as the CORINE (COoRdination Of INformation on the Environment) land cover maps provided by the European Environmental Agency only became available after 1990, while data collected at farm structure surveys conducted by the Austrian statistical agency were inconsistent over the years of assessment and turned insignificant when added to the estimation. For example, in some cases the forest cover reported for a district would be larger than the district itself. While certainly not perfect, we hope that our time-dummies have taken out some of these effects caused by structural change. Second, lack of data also concerns feeding practices and other wildlife management measures that we could not control for in the regression and are therefore either part of the fixed effects of the error terms. Third, the assumption of a linear relationship between highway density and harvest density may only be a rough approximation of the true effect and therefore should only be interpreted within the vicinity of the current observations.

4.4.3 Implications for Environmental Impact Assessment

We show that the impacts of highways on game species are complex, because they combine habitat loss and fragmentation effects, each of which dominate depending on whether highway density increases in a given district or in a neighbor district. Our study of recent highway EIAs revealed that certain stakeholder groups (particularly hunters) are intensively consulted, while others (e.g. forest owners, beneficiaries from forest ecosystem services such as recreationists or tourists, conservationists) have been consulted less. Our findings are very well in line with Beattie's 1995 statement that while EIAs are non-scientific value laden advocacy documents, they inform decisions and provide valuable information about public decisions. However, current EIA practice still lacks standardized base line data to enable objective and transparent assessments.

For future EIAs, we can formulate some specific recommendations based on the results of our study

1. Highways impact different species at different levels. Before any recommendation can be given in an EIA, there needs to be a clear strategy on which species are of interest in a given region. This would make the scoping procedure more efficient and allow more depth for the investigation of the relevant species. We provide a simple method for quantitative assessment of effects on popular game species

using readily-available data. For other species of interest, a similar method could be used to assess whether or not a certain project will have likely wider-ranging impacts on a species. While some EIAs collected data on annual harvest and wildlife diseases (i.e. ASFINAG, 2012b; Barbl, 2009; REVITAL ecoconsult, 2007), they did not go beyond reporting descriptive statistics and graphics.

2. Spatial effects are important. We therefore recommend widening the geographical focus of EIAs to consider how indirect effects of a highway project may influence neighboring areas. The spill-over effect (or telecoupling effect) is often ignored in the practice of EIAs due to administrative division. This includes possible positive abundance effects and their impacts on agriculture, forestry, and hunting. Assessments could be supported by developing and using quantitative modelling tools, such as the model applied in the current paper.

Finally, it has to be kept in mind that our findings can only be interpreted within the context of the available data. While the panel data allow us to control for district and time fixed effects, more sophisticated data such as population abundance estimates would allow for a more precise estimation of the Schaefer model parameters or a different, more elaborate model of population dynamics. Currently, there are efforts to establish wildlife monitoring programs in several regions of Austria, until these data are available for a longer time period, harvest data will be the best source to study population dynamics at the aggregate level. In addition, our theoretical model requires the assumption of a (long-term) equilibrium, which may be too restrictive in some cases. It has to be considered that this quantitative model presents a possibility to generate a systemized base line indicator as reference. Structured qualitative assessments and more sophisticated knowledge on habitat qualities will be necessary to connect trends to identify underlying causes and to explore other potential variables. In the end, an EIA does not take the decision about a project approval. Nevertheless, scientifically solid information can provide sound base line data and thereby make connected value judgements of decision makers more transparent.

4.5 Conclusions

The construction of fenced highways could affect the population of wild animals through habitat loss and landscape fragmentation, which might be positive or negative. For example, an increase in the number of fenced roads could reduce the number of wildlife-vehicle collisions. We use a spatial lag model and district aggregate game harvest data after 1968 to estimate how highway construction has impacted the population of red deer, roe deer and wild boar in Austria. We found that populations of roe deer and wild boar decrease with an increase in highway density in the same region. However, populations increase as highway density in neighbor districts is increased. These effects were even stronger for roe deer after Austria joined the EU and was obligated to conduct environmental impact assessments for highway projects. The results provide a baseline for improved environmental impact assessments in the context of future highway construction projects in Europe.

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Chapter 5

Conclusions, policy implications, and future research

Economic growth has led to substantial losses in biodiversity and environmental quality. In response, many countries have adopted environmental policies to reduce negative impacts of human economic behavior on the environment. But environmental policy remains hotly debated. While both advocates and opponents of environmental policy may agree that some form of public intervention is necessary to stop environmental decline, the question remains how to implement environmental policy in the most effective way. It is therefore important to improve the knowledge base of how environmental policies have worked in the past, to find defensible impact evaluation methods, and to warn from possible caveats of different policy implementation strategies. The estimation of policy impacts is non-trivial, and this dissertation addresses some of the complex issues in teasing out the effects of several environmental policies from complex economic-ecological systems. In this concluding chapter, we summarize our key findings, and provide some policy recommendations and ideas for future research related to each of our papers.

1. Payments for environmental services (PES) schemes for voluntary agri-environmental measures may be ineffective. Making them more effective would require a re-thinking of agri-environmental programs. For example, in the 2007-2013 programming period, the largest share of AEP funding in Austria went to a measure named UBAG, a measure that had no specific biodiversity goals but was rather designed as a low entry-barrier measure, making it susceptible to a hidden income transfer producing windfall effects rather than biodiversity improvements (the equivalent measure in the 2000-2006 program was named ”Grundförderung”, which is a bit more revealing). Designing future AEPs with specific ecological goals in mind (e.g. by examining the Annexes of the Habitats Directive for species and habitats of community interest) would make defining measurable goals and monitoring easier, and policy adjustments could be based on actual outcomes rather than political processes. Further, public expenditure could be better justified when being assessed by public auditing bodies.

   It is important to note that environmental impacts of an AEP depend on (1) the intensity of farming in an area and (2) on species ecology. Latent class analyses for important indicator species could help to investigate which agricultural characteristics of a region make PES lead to significant changes in habitat conditions. As more and better data on habitat conditions and wildlife population sizes become available, policies should adjust to better target those species and habitats of community interest. Thereby, funding can be shifted towards those regions where desired impacts are most likely. This could not only lead to a more effective use of subsidies, but also reduce costs for monitoring and evaluation.

2. As an alternative to the voluntary PES schemes studied in the first paper, the implementation of the Natura 2000 network of protected areas has followed a more targeted approach. In particular, based on specific lists of threatened species and habitats, EU member states were obliged to designate environmentally sensitive areas to the Natura 2000 network. Being an integrated conservation concept, this designation may have wide-ranging consequences to economic activity within protected sites. According to our study, Natura 2000 designation has negative effects on farmland rental prices. Our study goes beyond the classical economic analysis of prices by not only modeling the (hedonic) price function, but also modeling the political process that may have led to designation of Natura 2000 farmland. We argue that site designation may have been endogenous with land prices, and therefore standard regression analysis is inappropriate. This is because site designation was not always a pure top-down process, but involved stakeholder participation and influence of interest groups. Future investigations could study this phenomenon for a longer time frame (using panel data) and study whether adjustments to current funding levels have occurred (dynamic adjustment). Future policies could try to increase the market value of Natura 2000 sites, e.g. by promoting sustainable (eco-) tourism in protected sites or by helping farmers to gain additional revenues from their products through new and innovative marketing strategies.

   The current data availability on Natura 2000 farming is limited. Even agricultural censuses in Germany only collect the presence or absence of Natura 2000 subsidies for farmers as binary variables, but no data were available on the amount of farmland under Natura 2000 per se. This is particularly problematic for states that do not pay any subsidies, but where Natura 2000 conservation policies may be in place nevertheless, because it means that Natura 2000 farms in these states are not observed as such. We found similar issues when trying the analysis of paper 2 using 2009 data from the Farm Accountancy Data Network (FADN). For future agricultural censuses, we recommend to increase the detail in the assessment of Natura 2000 farming characteristics, by adding questions for area, payments, rented land and rental prices of Natura 2000 farmland. This would allow a direct within-farm comparison and eliminate many of the caveats we faced in our analysis.

3. The construction of highways has affected wildlife populations worldwide, and the main channels are habitat destruction and landscape fragmentation. Our results suggest that the impacts of habitat loss are always negative, but our findings, and findings in the literature have suggested that fragmentation may actually increase population sizes. Depending on the species, each of these effects may be associated with new challenges for hunters, farmers, and landowners. While all recent highway projects in Austria included wildlife EIAs, the quality of these was often questionable. We found key shortcomings in the applied methods and the consulted literature. For the improvement of future EIAs, we therefore recommend investing into the science-policy interface to improve

the diffusion of the latest research into EIA procedures. From an ecological perspective, habitat destruction and possible second-order effects seem to be the most important negative impacts on the wildlife populations, compared to fragmentation effects from highways. Therefore, reducing the destruction and degradation of habitats by minimizing land take, reducing the disturbance from noise through speed limits or noise barriers (although their effect on habitat quality is not well understood yet and will require further research (Shannon et al., 2016)), and providing substitute habitat could be effective policies to minimize the impact of highways on wildlife. Considering the wider geographical effects if highways will also be important in future highway EIAs.

Obviously our research has some limitations, both in the assumptions of the theoretical models and in the data used. Data availability is always a key issue when doing empirical research. Driven by the study subjects, this dissertation relies on secondary administrative data rather than self-collected primary data.

For the Austrian wildlife data, we were able to assemble a long time-series of district-level data. While this resolution is already quite high, it is not fine enough to model ecological mechanisms in detail. A first step to improve this shortcoming would be to provide wildlife harvest data at a lower administrative level (e.g. municipality level). A second, related issue is the availability of reliable and precise land-use data. For Austria, land-use data have been collected with the agricultural census in irregular intervals over the past 50 years, but these data are owner-based rather than place-based, meaning that a certain type of land use is attributed to the district of their owner than the district where land-use is actually taking place. We have found that these two districts often do not coincide, rendering them unsuitable to model habitat conditions. The CORINE project[1] shows some promise, but current temporal and spatial resolutions of the available data are not fine enough for well-founded ecological-economic modeling at a larger scale. Investing in capacities to produce high-quality land use monitoring maps could not only help to improve ecological and economic models, but will provide useful data for many other disciplines. Future research should use these data to relax some of our restrictive assumptions (e.g. the Schaefer hypothesis used in two papers).

In this dissertation, we studied three different implementations of environmental policy. It showed that not only the design and implementation of environmental policy are challenging, but also the evaluation of (side-) effects. While progress in the available data and the methods of analysis has been made, theoretical and empirical constraints make the choice of models for the analysis depend on value judgments by the researcher. Different methods and models may lead to different results when studying the same subject, but similar results based on different data and methods will strengthen the empirical evidence of a given relationship. We have tried our best to make our value judgments as transparent as possible. Although our data and methods certainly have weaknesses, the results of our studies are well-supported by the scientific literature. We hope our findings strengthen the scientific base for improving the effectiveness of future environmental policies.

[1] https://land.copernicus.eu/pan-european/corine-land-cover

Additional comments

During my time here in Göttingen, I had the pleasure to work with other authors on several other papers. My own estimated contributions to these papers are in brackets following the citations. Together with Ulrich Morawetz from BOKU Vienna, I have published two papers, one on the trail preferences of mountain bikers in Austria (Koemle & Morawetz, 2016) (90%), and the second one on measures against urban heat in Vienna (Morawetz & Koemle, 2017) (20-30%). With Elisa Giampietri, Xiaohua Yu and Adele Finco, I published a paper on the sustainability dimensions of farmers markets (Giampietri, Koemle, Yu, & Finco, 2016) (30-40%). With Feifei Sun and Xiaohua Yu, we published a paper on the relationship between air pollution and food prices in Beijing (Sun, Koemle, & Yu, 2017) (30%), and with De Zhou a paper on feed and hog markets in China (Zhou & Koemle, 2015) (20%). I further had the chance to present my work at workshops in Nanjing, China (2014), Kyoto Japan (2016 and 2017) and in Hanoi and Viet Tri, Vietnam (2018), which were great opportunities to extend my international experience. I thank all the involved persons for their cooperation, commitment, and support.

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