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Highway construction and wildlife populations: Evidence from Austria

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ABSTRACT

Fragmentation and destruction of ecosystems due to highways are key threats 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.

1. Introduction

The construction of highways diminishes resources for many wildlife species globally (Fahrig and Rytwinski, 2009; Forman and Alexander, 1998; Newbold et al., 2015; van der Ree et al., 2015; Völk and Glitzner, 2000; Völk and 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 (Baylis et al., 2016; Haddad, 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 and 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 et al., 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 and Greig, 2006; Masden et al., 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 et al., 2001), as well as lack of competence and training of the personnel (Zhang et al., 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

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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.

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.

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 et al., 2015). Second, highways through animal habitat increase light and noise pollution, air pollution through gas emissions (Huang et al., 2009) and dust (Nanos and 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 et al., 2016; Laurian et al., 2008; Rost and 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 et al., 2001; van der Ree et al., 2015).

Third, as a further effect, highways may lead to increased development efforts in their vicinity (Selva et al., 2015). In the literature, roads have been identified as being one important determinant of deforestation (Chomitz and Gray, 1999; Deng et al., 2011). Mothorpe et al. (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, Fig. 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 et al., 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 and 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 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 and Reimoser, 2010), while paratuberculosis has been found in red deer (Fink et al., 2015; Schoepf et al., 2012) and roe deer (see Duscher et al. (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. Fig. 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 and Fahrig, 2004).

Finally, separating two habitats by a highway may decrease the intra-and inter-specific competition effect that a species experiences (Fahrig, 2017). Separation of habitats could lead to a sudden decrease in interference competition (Begon et al., 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 and Rytwinski, 2009; Liao et al., 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.

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 Fig. 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 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

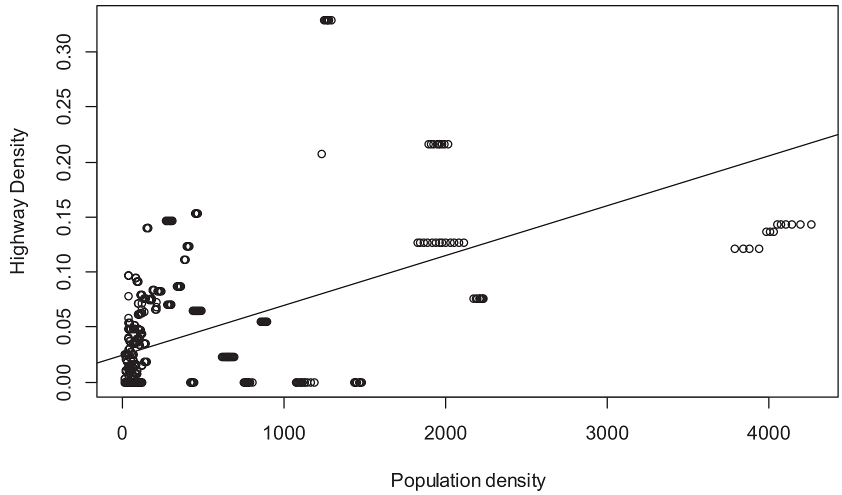

Fig. 1. Density of highways plotted against human population density (years 2002–2014).
Source: Statistik Austria

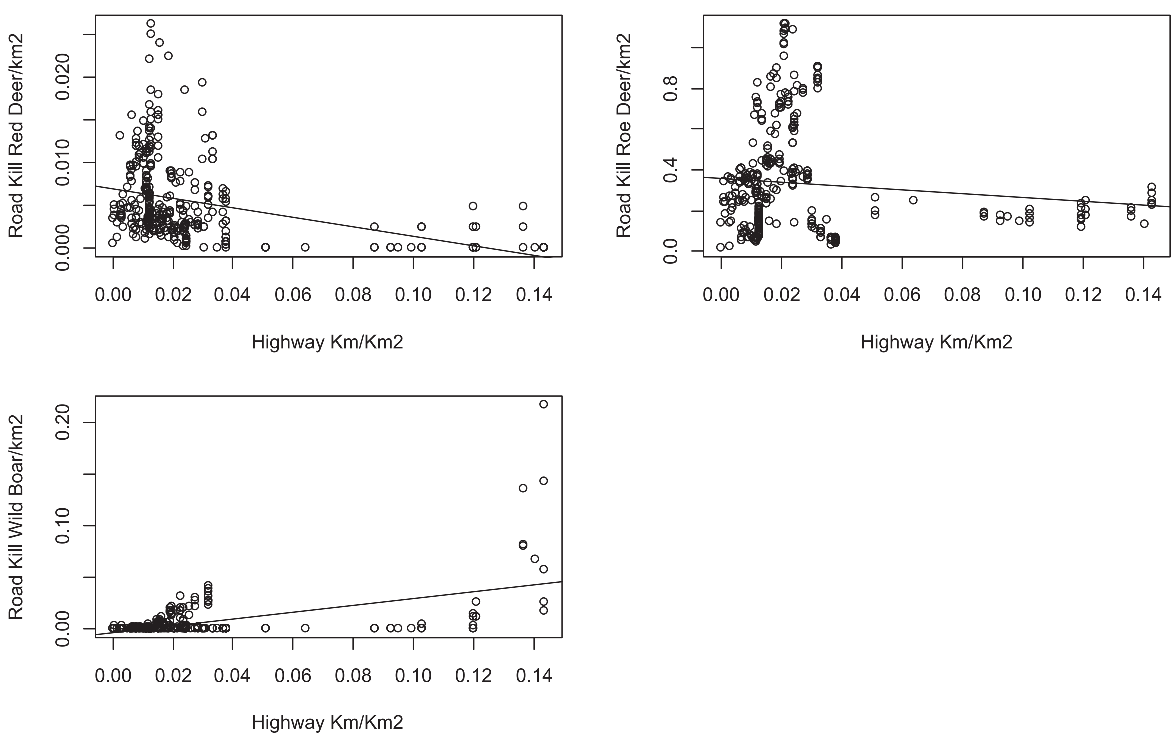

Fig. 2. Road kill per km^2 of red deer, roe deer and wild boar plotted against the density of fenced roads in Austria (province level 1968–2014).
Source: Statistik Austria

highways are in operation, 26 km are currently in construction, and 31 km are planned (bmvit, 2016). Fig. 4 shows the current spatial distribution of highways in Austria.

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 Fig. 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, 2001a). 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 and Kušta, 2017; Kušta et al., 2017). According to Bauer (2001a) 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 its home (Bauer, 2001b). 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 (Fig. 5).

Wild boar today is mostly present in the eastern parts of Austria (Fig. 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

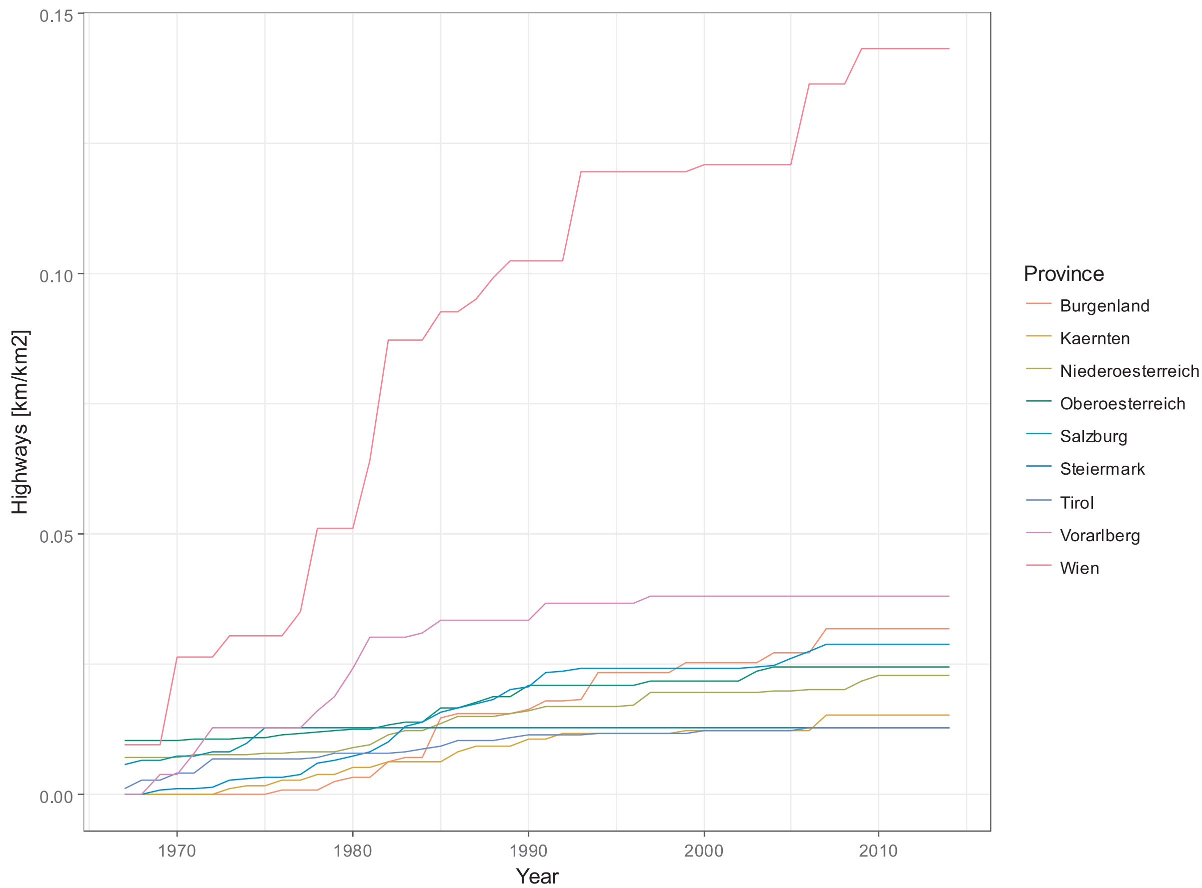

Fig. 3. Changes in cumulative highway density in the nine provinces of Austria.
Source: ASFINAG 2012, own calculations

Fig. 4. Current (2016) highways in Austria.
Source: OpenStreetMap

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

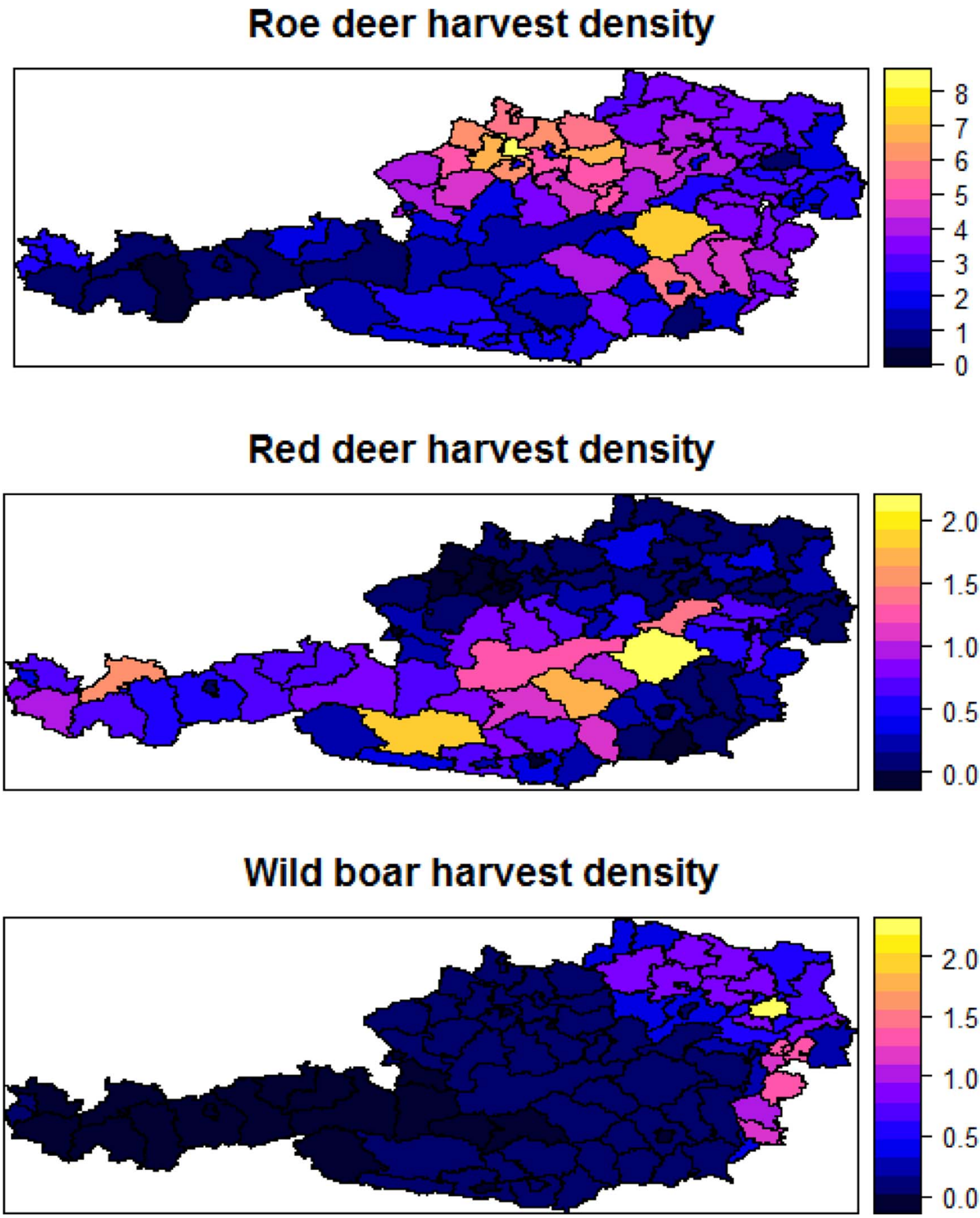

Fig. 5. Distribution of the average 1968–2014 harvest density (individuals/km^2) of red deer, roe deer, and wild boar in Austria.
Source: Statistik Austria

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 et al., 2016; Völk and 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.

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 et al., 2013).

In Austria, EIAs have been mandatory after 1993, and the first highway project subject to EIA was started in 1996.[1] 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 et al., 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 and Nooteboom, 1999).

Jaeger (2015, p. 34) 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 modelling 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 (e.g. Glasson et al., 2013), 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.

3. Data and methods

3.1. Theoretical framework

The well-known Schaefer model describes population dynamics as a logistic growth model minus annual harvest (Conrad and Clark, 1987, p. 68; Schaefer, 1957)

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

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

$$H_t = qN_tE_t \tag{2}$$

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 (2)

$$H_t^* = qKE_t - \frac{q^2K_t}{r_t}E^2 = \left(q - \frac{q^2E_t}{r_t}\right)K_tE_t \tag{3}$$

or more generally, the optimal harvest is

$$H_t^* = H(K_t, r_t, E_t) \tag{3a}$$

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, Eq. (3a) could be rewritten as

$$H^* = f(r(D, ND), K(D, ND), E)$$

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

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

and

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

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.

[1] S1 Wiener Südrand Schnellstraße, Schwechat – Vösendorf.

$\frac{dH^*}{dND}$ 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.

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 et al., 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

$$H_{i,t} = c_t + f_i + \gamma_1 Temperature_{i,t} + \gamma_3 Precipitation_{i,t} + \Sigma_k \gamma_{3,k} Competitor_{k,i,t} + \gamma_{41} D_{i,t} + \gamma_{51}[WND_{i,t}] + \gamma_{42} D_{i,t} I95_{i,t} + \gamma_{52}[WND_{i,t}] I95_{i,t} + \gamma_6 H_{i,t\text{-}1} + \gamma_7[WH_{i,t}] + \gamma_8 licenses_{i,t} \quad (4)$$

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 and 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 et al., 2013; Bivand and 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 over-identification, and the Arellano-Bond test for autocorrelated residuals.

3.3. Data

The data for estimation Eq. (4) were collected from several sources. Descriptive statistics are presented in Table 1.

- 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.

Table 1
Descriptive statistics.

Variable	Unit	Observations	Mean	Std. Dev.	Min.	Max.
Roe deer density	harvest/km^2	4560	2.98	1.91	0	12.89
Red deer density	harvest/km^2	4560	0.37	0.49	0	4.88
Wild boar density	harvest/km^2	4560	0.24	0.57	0	5.47
Highway density	km/km^2	4560	0.02	0.04	0	0.33
Average Temperature	1/10 °C	4560	76.84	15.62	24.49	117.63
Annual Precipitation	mm	4560	925.68	215.35	320.63	1894.52
Hunting licenses	licenses/km^2	4560	1.30	0.33	0.08	3.26

- 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 map tools (Bivand and 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/km^2) indicator (Forman and 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.

4. Results and discussion

4.1. Results

The results of three Arellano-Bond models are shown in Table 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.

4.2. 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.

Table 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.

Model Dependent variable	(1) Roe deer density	(2) Red deer density	(3) Wild boar density
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 Highways 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 density	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 density		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 density			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)
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	4370	4370	4370
Number of districts	95	95	95
Sargan Test Chi^2 (p-value)	2836.85	2974.38	84.58
	(0.8243)	(0.1914)	(1.000)
1st order Arellano-Bond Test (p-value)			−3.2534
			(0.0011)
2nd order Arellano-Bond Test (p-value)			0.99273
			(0.3208)

Standard errors in parentheses.
*** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$.

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 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 et al., 2004; Jirinec et al., 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.

4.3. 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.

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.

Conflict of interest statement

The authors declare no conflict of interest.

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