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Predicted impacts of transport infrastructure and traffic on bird conservation in Swedish Special Protection Areas

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Abstract

The ecological impacts of roads and railways extend into the surrounding landscape, leading to habitat degradation and reduced wildlife densities within an area that is considerably larger than the actual road or railway corridor. For birds, an extensive meta-analysis has identified an average of 20% density reduction within 1 km from the infrastructure. I investigated to what extent this density reduction can be expected to compromise the habitat quality and conservation value of Swedish Natura 2000 areas designated for the protection of birds (Special Protection Areas; SPAs). The majority (63%) of Swedish SPAs are, to some extent, found within this 1 km road/railway effect zone (REZ). The total overlap between SPA and REZ is approximately 126,000 ha or 4.2% of the country's SPA area. There are, however, large differences amongst bio-geographical regions. In the southern (continental) and coastal regions combined, 25.8% of the total SPA area fall within REZ, representing an estimated 4–7% reduction in bird abundance within SPAs. The probability of overlap with REZ is higher for larger SPAs. However, the proportion of overlap is higher for smaller SPAs and, accordingly, smaller sites can be assumed to experience a greater impact from transport infrastructure and traffic. The impacts on Natura 2000 sites are particularly concerning as this network of protected areas is a cornerstone for maintenance and restoration of biodiversity within the EU. I recommend placing a stronger emphasis in the management of Natura 2000 sites on the threats to wildlife conservation caused by transport infrastructure and traffic. Special attention should be paid to sites with a large overlap with the REZ and sites hosting particularly vulnerable taxa or habitats. Infrastructure owners and managers should make their best efforts to minimise and compensate for the negative impacts of roads and railways and associated traffic in SPAs and other protected areas.

Keywords
Effect zone, Natura 2000, railway, road, Sweden

Introduction

Ecological impact of transport infrastructure

Infrastructure development is recognised as one of the significant drivers of global biodiversity loss and, with increasing traffic and expanding infrastructure networks worldwide, the pressure on biodiversity is expected to increase in coming decades (EEA-FOEN 2011, EEA 2012, OECD 2012). The impacts of transport infrastructure on wildlife are well described (Forman et al. 2003, van der Ree et al. 2015) and include loss of habitat, traffic casualties, creation of physical barriers, disturbance by noise, light and other visual cues, spread of chemicals, dust and alien species, changes in hydrology and microclimate and accidental spills. Most of these impacts extend into the surrounding landscape, leading to degradation and fragmentation of habitats and, for some species, to restricted movements, increased mortality and avoidance of a zone around the infrastructure (Forman et al. 2003, EEA-FOEN 2011, van der Ree et al. 2015, Tulloch et al. 2019).

Due to these impacts, the population densities of many wildlife species are reduced within a distance from larger infrastructures (Rytwinski and Fahrig 2015). For example, the population densities may be reduced up to about 1 km distance for birds (Forman and Deblinger 2000, Forman et al. 2002, Benítez-López et al. 2010) and anurans (Eigenbrod et al. 2009) and up to 5 km for mammals (Benítez-López et al. 2010). Not only large infrastructure but also minor and unpaved roads may have a considerable impact on some species (e.g. van Langevelde and Jaarsma 2009, Benítez-López et al. 2010, Shanley and Pyare 2011). Based on an extensive meta-analysis, Benítez-López et al. (2010) showed that the mean bird and mammal abundance in an effect zone around infrastructures is reduced by 20–30% and with an increasing reduction with proximity to the infrastructure (Fig. 1).

Accordingly, in regions with dense infrastructure networks, large natural areas may be situated within this effect zone and therefore may be impoverished in species sensitive to traffic and transport infrastructure. For example, in the United States, the road effect zone covers 15–22% of the total land area and more than 60% of some particularly exposed biomes, such as coastal regions and river basins (Forman 2000, Riitters and Wickham 2003). In Spain, a country with intermediate road density with European standards, it is expected that reduced bird densities, due to transportation infrastructure, will affect 49% of the country and lead to reduced mammal densities in as much as 96% of the country (Torres et al. 2016). Some habitats of particular importance to biodiversity in Europe, such as wetlands, semi-natural grasslands and temperate broad-leaved forest, may be disproportionately affected by roads because of the landscape structure (Helldin et al. 2013, Karlson and Mörtberg 2015, Torres et al. 2016). Disturbances (noise and visual cues) tend to be stronger and extend further

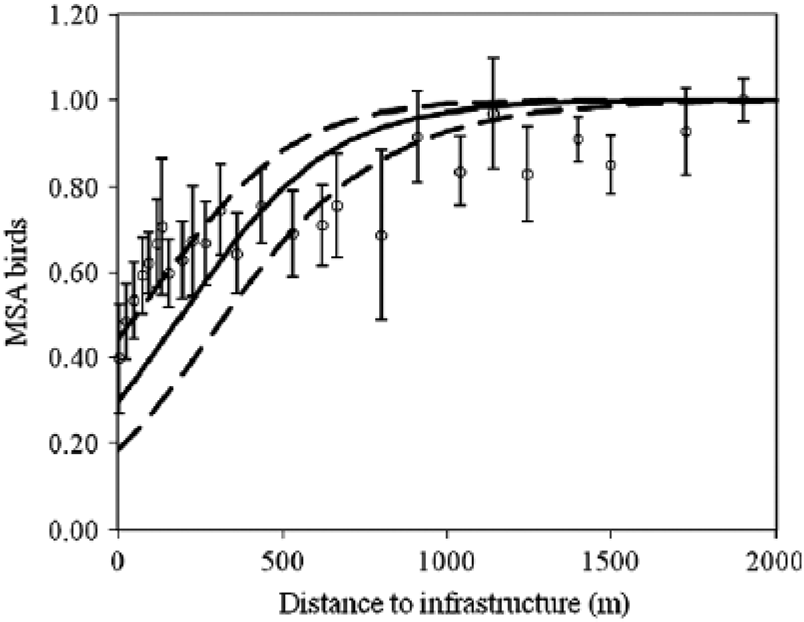

Figure 1. Mean species abundance (MSA) of birds as a function of distance to infrastructure (logistic regression). Open dots represent the pooled results of a meta-analysis per distance interval ±SE. The solid black line denotes the estimated curve for the decline of MSA in proximity to infrastructure; dashed lines are the 95% upper and lower limits of the confidence bands of the curve. Figure from Benítez-López et al. (2010).

from the road in open habitats as compared to forest (Forman and Deblinger 2000, Reijnen and Foppen 2006). There is growing concern globally about the impact of roads and traffic on wildlife populations in protected areas and other biodiversity hotspots (Forman and Deblinger 2000, Ament et al. 2008, Selva et al. 2011, Laurance and Balmford 2013, Bager et al. 2015, Gadd 2015, Seshadri and Ganesh 2015).

Integrity of the Natura 2000 network

The Natura 2000 network of protected areas is a key tool in the maintenance and restoration of biodiversity within the European Union (EU). The network consists of Special Protection Areas (SPAs), designated according to the Birds Directive and Special Areas of Conservation (SACs), designated according to the Habitats Directive (EEC 1992, EEC 2010). Under these directives, the network is supposed to provide protection for vulnerable wildlife and habitats. One important motivation for designating Natura 2000 sites, particularly in coastal and other lower-elevation areas, is the protection against negative impacts of urbanisation and infrastructure development (EEA 2012). There are currently more than 26,400 sites in the Natura 2000 network, accounting for ca. 18% of EU's land territory.

The network is, however, biased toward highland areas, and lowland areas are under-represented (Oldfield et al. 2004, Maiorano et al. 2007). In addition, the average size of the Natura 2000 sites is quite low and particularly so in lowlands (Maiorano et al. 2007, Gaston et al. 2008, EEA 2012). As smaller sites are more susceptible to pressure from land use and human activities surrounding them, major concerns are expressed about the capacity of existing protected areas to maintain their biodiversity values (Chape et al. 2005, Gaston et al. 2008, Maiorano et al. 2008, Kati et al. 2015).

Though many assessments for the effectiveness of the Natura 2000 network in protecting species have been reported in recent years, a vast majority of these relied on rather static population data, such as species' geographic distribution, species presence/absence or habitat suitability (e.g. Araújo et al. 2007, Maiorano et al. 2007, 2015, Sánchez-Fernández et al. 2008, Iojă et al 2010, López-López et al. 2011, Gruber et al. 2012, Albuquerque et al. 2013, D'Amen et al. 2013, Lison et al. 2013, Rubio-Salcedo et al. 2013, Trochet and Schmeller 2013) and, accordingly, were designed to assess the ecological representation of the network rather than tracking changes in population densities due to environmental impacts. In view of the many negative population trajectories reported for both avian and non-avian species in the EU (EEA 2015), it appears necessary to analyse the ecological functionality of the Natura 2000 network with regard to pressures both within and outside the designated areas, but few studies have taken this course.

Frequency of transport infrastructure within Natura 2000 sites was investigated by Tsiafouli et al. (2013), showing that as a European average, roads are present in 29% of the sites, with a higher frequency in the countries in the south. A preceding study of Greek Natura 2000 sites, by Votsi et al. (2012), showed that 85% of sites are bisected by roads. Insufficient functional connectivity of the Natura 2000 network has been reported by Gurrutxaga et al. (2011) and Opermanis et al. (2012), suggesting that dispersal barriers exist between sites, for example in the form of large roads (Gurrutxaga et al. 2011). With regard to disturbance, the European Environmental Agency estimated that almost 20% of Natura 2000 areas are presently adversely affected by high levels of environmental noise, largely owing to major transport infrastructure (EEA 2016). In an assessment of the impact on Natura 2000 sites of major traffic arteries, planned or under construction as part of EU's TEN-T framework, Byron and Arnold (2008) estimated that 379 SPAs (8% of sites) and 953 SCIs (4% of sites) would be adversely affected by these new traffic arteries, with potential effects also on the coherence of the Natura 2000 network.

Aim of the study

Sweden is one of the European countries that is least fragmented by transport infrastructure and built-up areas (EEA-FOEN 2011) and the physical impact of infrastructure is generally not well acknowledged in Swedish nature conservation. It was not until recently that national status reports for biodiversity have described the threats from infrastructure and traffic on species conservation (Bernes 2011, Naturvårdsverket 2015a) and the current national conservation action plan contains few requirements for infrastructure managers (Naturvårdsverket 2015b). Biodiversity is generally insufficiently described in impact assessments for transportation infrastructure plans and projects (Wärnbäck 2013, Karlson et al. 2014). Few, if any, management plans for protected areas address the full array of potential ecological impacts of transport infrastructure on the areas' conservation status or management (Helldin and Tytor 2017). This ignorance is not unique for Sweden, as it appears to be largely similar in most EU member states (Tsiafouli et al. 2013, EEA 2015; but see, for example, Selva et al. 2011, Votsi et al. 2012).

In order to illustrate and highlight the impacts of infrastructure on protected areas in particular, I estimated the frequency and proportion of Swedish SPAs situated within the predicted effect zone for birds around existing larger transport infrastructure (roads and railways) and hence cannot be expected to reach their full conservation potential due to infrastructure impacts. I included only SPAs in the estimation, i.e. areas designated specifically for the protection of birds, because the effects of roads and railways on birds are well described in literature and apparently can impact the majority of bird species (Reijnen and Foppen 2006, Benítez-López et al. 2010, Rytwinski and Fahrig 2015). I assessed the conservation value of SPAs that is lost due to infrastructure in terms of reduced predicted bird abundance. Due to the large geographic variation over the country in density of the infrastructure network and proportion of area within an SPA, I separated the analyses between biogeographical regions. I tentatively explored the association between the infrastructure impact on an individual SPA and its size and dominating habitats. In this paper, I present the results of these analyses and propose improvements for the management of Swedish SPAs.

Methods

A map of the effect zone for birds around the existing larger Swedish roads and railways (REZ in the following) was produced. I used a standardised effect distance of 1 km from the infrastructure, following the results from a meta-analysis presented by Benítez-López et al. (2010). To my knowledge, this is the most comprehensive analysis of infrastructure impacts on bird densities, including 49 bird datasets and 201 bird species. As most studies in the meta-analysis were conducted in biomes that occur in Sweden, i.e. taiga, temperate broadleaf forest or alpine/tundra (39 of the 49 datasets) and on species that occur in Sweden (105 of the 201 species), I judged the results to be relevant to a Swedish perspective. The results have also previously been applied to assess the impacts of the road network on birds in Sweden (Karlson and Mörtberg 2015) and Europe (Torres et al. 2016). I consider the assumption of a 1 km effect distance to be conservative because i) individual studies in the meta-analysis indicated reduced bird populations at larger distances and ii) impacts at greater distances may not necessarily result in direct population declines, but yet be of ecological significance. Infrastructure data were obtained in December 2015 from Open Street Map (http://openstreetmap-data.com), using only the following road classes: primary road, secondary road, tertiary road, motorway, trunk road and railway (thus excluding minor roads for which ecological effects are less well known).

The REZ map was overlaid with all Swedish SPAs to calculate the area and proportion of each site situated within REZ. The area and habitat distribution of SPAs were obtained in December 2015 from European Environment Agency's Natura 2000 database (http://www.eea.europa.eu/data-and-maps/data/natura-2000-eunis-database). SPAs were separated depending on biome (based on a combination of the Natura 2000 database and Global Biomes data from the CIESIN; http://sedac.ciesin.columbia.edu/data/set/nagdc-population-landscape-climate-estimates-v3/maps?facets=theme:climate) and proximity to coast (data on coastline obtained in December 2015 from Open Street Map; http://openstreetmapdata.com) on the following terms:

- *continental region:* sites with >50% of the area within EU continental region,
- *mixed-forest region:* sites with >50% of the area within EU boreal region and in CIESIN broadleaf and mixed-forest region,
- *boreal region:* rest of sites within EU boreal region but with no part within EU alpine region or
- *alpine region:* sites with at least some part of the area within EU alpine region

in combination with

- *coastal:* mainland sites with at least some part within 20 km from coast of mainland Sweden (including mainland islands Öland and Gotland) or
- *inland:* the rest of mainland sites.

The alpine region in Sweden is only inland, i.e. no alpine coastal sites exist. As only two continental sites are inland, all continental sites were pooled in one region. In addition, an off-coastal region was formed including all sites with no contact with mainland Sweden, irrespective of terrestrial biome. Hereby, a total of seven biogeographical regions were obtained (Table 1 and Fig. 2).

To assess the bird conservation value of SPAs that is lost due to infrastructure, I used an average of 20% (C.I. 12–33%) reduction of bird abundance within the 1 km effect distance from the infrastructure, as indicated by the results presented by Benítez-López et al. (2010; Fig. 1). GIS analyses were conducted using ArcGIS version 10.2 and QGIS version 2.6. To explore the differences between biogeographical regions in the proportion of SPA area within REZ, an ANOVA with Tukey´s post-hoc test was conducted, with region as explanatory variable. To explore how the degree of impact on an SPA is associated with its size and dominating habitat, two different analyses were performed for each region. For the probability for an SPA to overlap to any degree with REZ (response variable either 0 or 1), a generalised linear model with a logit link function (logistic regression) was used, and for the proportion of overlap with REZ (i.e. a response variable from >0 to 1), a beta-regression model (Ferrari and Cribari-Neto 2004) was used, including only sites with overlap. In both types of models, explanatory variables were 1) SPA size, 2) proportion of forest habitat, 3) proportion of wetlands and 4) proportion of agricultural land and grasslands. SPA size was log-transformed to improve normality and all variables were standardised to make parameter estimates

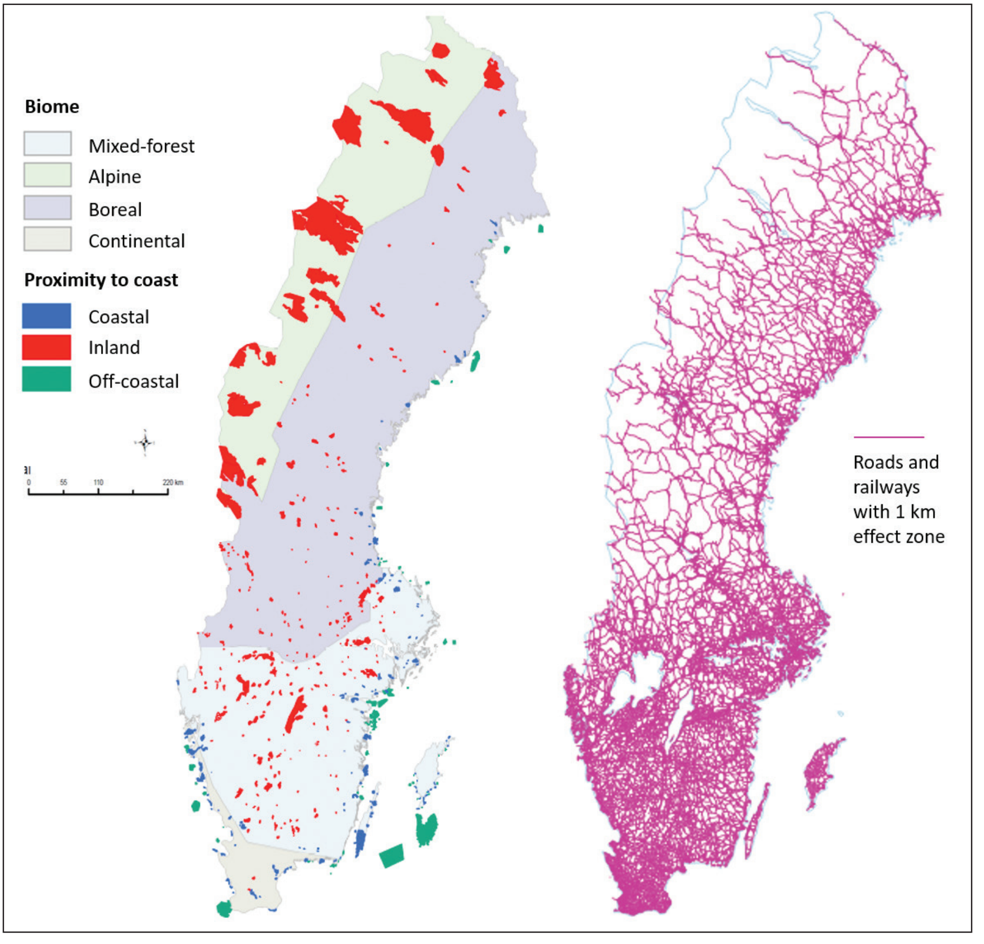

Figure 2. Special Protection Areas (SPAs) in Sweden divided by biogeographical region (left map) and larger roads and railways with predicted effect zone (REZ; right map).

comparable. Model selection was based on AIC and the final models were the ones with a combination of explanatory variables resulting in the lowest AIC. Statistical analyses were conducted using programme R version 3.4.2.

Results

The overlay of SPAs and REZ showed that 339 of Sweden´s 538 SPAs (63%) have at least some part and 123 (23%) have most of their area within REZ (Table 1). In terms of area, a total of approximately 126,000 ha or 4.2% of the total SPA area in the country lies within REZ.

However, national level figures on the impacted area give a crude picture, as the results pointed to large differences amongst the biogeographical regions (Table 1). The proportion of SPA within REZ differed among regions (F = 13.8, p <0.001), with SPAs in the alpine and off-coastal regions having a significantly smaller proportion of their area in REZ than the other regions, and boreal inland SPAs having a significantly smaller proportion than continental SPAs. Alpine and off-coastal areas have a number of large SPAs and hold most of Sweden´s total SPA area, but have sparse networks of large (terrestrial) transport infrastructure and, accordingly, they reduce the national average. Continental, mixed-forest coastal and boreal coastal regions, however, are comparably more impacted; in these three regions combined, a total of 46,046 ha or 25.8% of the total SPA area fall

Table 1. Predicted impacts on SPAs. Number, area and proportion of Swedish SPAs within an effect zone of 1 km from larger transport infrastructure (REZ) and predicted total reduction in bird abundance (with 95% confidence interval) due to the effects. Results are given for the entire country and by biogeographical region. Detailed results for each SPA are provided in appendix available online at http://triekol.se/earlier-work/infrastructure-impacts-on-protected-areas/.

Region	Total no. of SPAs	Total area of SPA (ha)	No. of SPAs affected to		Total area of SPA in REZ (ha)	Proportion of total SPA in REZ (%)	Reduction in bird abundance (% with C.I.)
			>0%	>50%			
Continental	41	53,331	41	18	19,202	36.0	7.2 (4.3–11.9)
Mixed-forest coastal	94	103,064	71	24	20,865	20.2	4.0 (2.4–6.7)
Mixed-forest inland	161	258,804	119	43	34,622	13.4	2.7 (1.6–4.4)
Boreal coastal	22	21,712	15	8	5,968	27.5	5.5 (3.3–9.1)
Boreal inland	130	162,924	63	30	13,122	8.1	1.6 (1.0–2.7)
Alpine (only inland)	26	1,984,005	16	0	28,242	1.4	0.3 (0.2–0.5)
Off-coastal	64	415,308	14	0	3,913	1.0	0.2 (0.1–0.3)
All Sweden	**538**	**2,999,149**	**339**	**123**	**125,946**	**4.2**	**0.8 (0.5–1.4)**

Table 2. Variables associated with road/railway effect zone (REZ) overlap with Swedish SPAs divided by biogeographical region. Values given are mean estimates of coefficients of logistic regressions and beta-regressions with standard errors (SE) and probabilities (P). Values are only given for variables that were included in the final model. The right column gives the variation explained by the final model.

Region	Log(size)		Forested area		Wetland		Agri & grassland		
	Estimate (SE)	*P*	Estimate (SE)	*P*	Estimate (SE)	*P*	Estimate (SE)	*P*	
Logistic regression (probability of overlap)									**Explained deviance**
Continental †	–	–	–	–	–	–	–	–	–
Mixed-forest coastal	0.45(0.27)	0.094							0.03
Mixed-forest inland					0.49(0.21)	0.019	0.91(0.38)	0.016	0.07
Boreal coastal			-1.06(0.57)	0.065					0.16
Boreal inland			-2.61(0.75)	<0.001	-2.06(0.70)	<0.001			0.21
Alpine (only inland)	1.31(0.64)	0.039			1.23(0.64)	0.056			0.23
Off-coastal	1.07(0.40)	0.008			0.79(0.32)	0.015			0.18
Beta-regression (proportion of overlap)									**Pseudo R^2**
Continental ‡	–	–	–	–	–	–	–	–	–
Mixed-forest coastal	-0.56(0.15)	<0.001					0.44(0.15)	0.002	0.26
Mixed-forest inland	-0.64(0.12)	<0.001					0.47(0.11)	<0.001	0.40
Boreal coastal	-0.85(0.34)	0.013							0.36
Boreal inland	-0.56(0.16)	<0.001	-0.48(0.16)	0.004			0.43(0.16)	0.009	0.41
Alpine (only inland)	-0.78(0.11)	<0.001			-0.38(0.17)	0.022			0.50
Off-coastal	-1.09(0.24)	<0.001					0.40(0.19)	0.036	0.38

† Could not be analysed as all sites had value 1, i.e. overlap to some degree with REZ.
‡ No variable contributed to the final model; i.e. the model including only the intercept had the lowest AIC.

within REZ. In terms of bird conservation value, present infrastructure impacts are predicted to cause a 4–7% reduction in bird abundance in SPAs in the three most impacted regions (continental, mixed-forest coastal and boreal coastal) and an average of 1% reduction when all SPAs in the country are taken into consideration (Table 1).

The overlap models with the lowest AIC included SPA size or habitat composition in all regions except the continental (Table 2). As the most strongly emergent pattern, larger SPAs appear to have a higher probability of overlap with REZ (in three regions), but a lower proportion of overlap with REZ (in all regions except continental). Another pattern, however less emergent, is that the overlap with REZ is larger in SPAs with more agricultural land and grasslands (in four regions) and smaller in SPAs with more forest habitat (in two regions). The pattern regarding overlap with SPAs with more wetland is inconclusive, and in general, only a minor part of the variation in overlap could be explained by SPA size and dominating habitat.

Discussion

The results indicate that a significant proportion of Swedish SPAs, both in terms of area and number of sites, lies within a predicted effect zone for birds around present larger transport infrastructure (REZ) and therefore can be expected not to reach their full potential as a bird habitat. The reduction due to transport infrastructure impacts may not be dramatic when seen in the country as a whole, with only around 4% of the total SPA area affected, corresponding to ca. 1% reduction in predicted bird abundance within SPAs. However, for more urbanised parts of the country, with a denser infrastructure network, the predicted impact and reduction is nearly an order of magnitude larger and may well be one of the main factors determining bird abundance in protected areas. This is the case in the southern (continental) and coastal regions of the country, where the urbanisation and landscape fragmentation is at a similar level to that of most western and central European countries (EEA-FOEN 2011).

At the level of individual SPAs, smaller sites tend to have a higher proportion of overlap with REZ and, accordingly, can be assumed to experience a greater impact than larger sites. This pattern emerges in all biogeographical regions, except the continental where most SPAs are small and indeed impacted to a large degree. In effect, the larger impact on smaller sites amplifies the bias against area protection in the lowlands, i.e. the southern and coastal regions. This is in line with general concerns previously expressed about the small size of many protected areas in Europe and about the impacts from transport infrastructure, traffic and other urban development in the landscape surrounding them (Shafer 1995, Gaston et al. 2008, Maiorano et al. 2008, Kati et al. 2015). However, as indicated by Helldin and Tytor (2017), these concerns are not well expressed in the management plans for Swedish SPAs and appear particularly underestimated in the regions where the impacts are the largest.

The functions of predicted bird abundance and distance to infrastructure described by Benítez-López et al. (2010) have previously been applied twice to assess the ecological impact of an infrastructure network at a larger geographical scale. Karlson and Mörtberg (2015) presented an assessment of the impacts of roads on habitats of high diversity value in Sweden (irrespective of their protection status), concluding that natural grasslands and southern broadleaved forest are likely to be particularly impacted; on a national level, 13–19% and 16–24%, respectively, of the total areas of these habitats are found within the predicted road effect zone for birds. Torres et al. (2016) estimated a 19.0% (CI: 9.6–25.6%) reduction in national bird numbers due to transport infrastructure for Spain, when considering all land, not only protected areas. They too concluded that some habitats (most notably farmland and maritime wetlands) might be disproportionately affected by transport infrastructure. In relation to these previous assessments, the present study is unique in that it points out the impacts specifically on protected areas, i.e. areas where nature conservation should be a top priority.

The assessment was aimed at providing a general picture and was therefore simplified in several respects. A fixed-width REZ is less realistic, as the actual effect depends on the local context, such as the habitat distribution, topography, species and ecological processes involved (Forman and Deblinger 2000, Riitters and Wickham 2003, Biglin and Dupigny-Giroux 2006), or the road characteristics (e.g. Reijnen and Foppen 2006, Rytwinski and Fahrig 2015). Additionally, the greater reduction in bird densities near the infrastructure within REZ (Fig. 1) provides an opportunity to assess the decline in bird abundance within the effect zone in individual sites in greater detail than conducted here (e.g. Torres et al. 2016, Tulloch et al. 2019).

Furthermore, the analysis of impacts on different habitats within SPAs was rather coarse, since the Natura 2000 database does not provide habitat maps. Therefore, I could not explore to what degree EU priority habitats (habitat types of community interest; EEC 1992) are distributed disproportionately within REZ.

Because the study was focused on transport infrastructures in terrestrial environments (roads and railways), the low predicted impact on off-coastal protected areas is not surprising. A corresponding analysis of the frequency of shipping and proximity to marine fairways would be of relevance to assess traffic impacts in marine reserves. Such an analysis would, however, need a different approach than the one here described, as the ecological effects of shipping are not yet comprehensively described and effect distances are less well established (Pirotta et al. 2019).

Implications for the management of protected areas

The present study underlines the concern about the impact of transport infrastructure on wildlife in protected areas in general (Forman and Deblinger 2000, Ament et al. 2008, Selva et al. 2011, Tsiafouli et al. 2013) and in the smaller areas in particular (Shafer 1995, Maiorano et al. 2008). Following article 4 of the Birds Directive, EU Member States must take appropriate steps to avoid habitat deterioration and significant species disturbance within SPAs (EEC 2010). Accordingly, a stronger emphasis on keeping natural areas free from the impacts of heavy traffic and new roads and railways has been proposed for conservation and transport planning (Selva et al. 2011, 2015, Laurance and Balmford 2013, IENE 2015).

Management plans for Natura 2000 sites should better acknowledge and consider the threats to wildlife conservation caused by both present transport infrastructure and new development projects in and near sites (Cortina and Boggia 2014, Helldin and Tytor 2017). Assessments of the effectiveness of individual Natura 2000 sites in maintaining biodiversity should include monitoring of population density and demography of species of special conservation concern (Gaston et al. 2008).

In order to correctly address the impacts from transport infrastructure, managers of Natura 2000 sites should conduct more detailed assessments of REZ in the respective site, based on habitat maps, species occurrences, and local road or railway characteristics (traffic, corridor width, etc.), to serve as a basis for setting priorities in conservation planning and action. Such site specific assessments could take into account the greater reduction in population density in close proximity to the transport infrastructure, and impacts on some wildlife species extending further than the 1 km used in the current study (Reijnen and Foppen 2006, Benítez-López et al. 2010). For example, Tulloch et al. (2019) describe a method to produce site- and taxon-specific REZs, incorporating local biodiversity distribution and expert opinion on the impacts on different taxa. Special attention to road effects should be paid in protected areas with a large overlap with REZ, in areas hosting particularly vulnerable taxa, and in areas with pronounced impacts on EU priority habitats.

Conservation authorities should ensure that infrastructure owners and managers make their best efforts to minimise the negative impact of nearby roads and railways and related traffic. Technical mitigation of impacts of transport infrastructure on birds could include preventing bird-vehicle collisions (e.g. with flight diverters), planning the timing of infrastructure maintenance and construction work to avoid particularly sensitive periods, providing crossing structures, and reducing noise and visual impacts through walls, berms or adapted paving. Such road mitigation measures for birds have been implemented on a few sites in Sweden, but are still lacking over the vast majority of roads and railways (Trafikverket 2017), and moreover, are not always considered in new construction projects. Traffic calming, speed reduction, or road closure (permanent or temporary) would also provide reductions in road mortality, disturbance, and barrier effects. Finally, compensatory measures such as habitat improvement or additional area protection could reinforce remnant populations and restore vital ecological processes.

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