



## Systematic review

# The impacts of roads and other infrastructure on mammal and bird populations: A meta-analysis

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## ABSTRACT

Biodiversity is being lost at an increased rate as a result of human activities. One of the major threats to biodiversity is infrastructural development. We used meta-analyses to study the effects of infrastructure proximity on mammal and bird populations. Data were gathered from 49 studies on 234 mammal and bird species. The main response by mammals and birds in the vicinity of infrastructure was either avoidance or a reduced population density. The mean species abundance, relative to non-disturbed distances (MSA), was used as the effect size measure. The impact of infrastructure distance on MSA was studied using meta-analyses. Possible sources of heterogeneity in the results of the meta-analysis were explored with meta-regression.

Mammal and bird population densities declined with their proximity to infrastructure. The effect of infrastructure on bird populations extended over distances up to about 1 km, and for mammal populations up to about 5 km. Mammals and birds seemed to avoid infrastructure in open areas over larger distances compared to forested areas, which could be related to the reduced visibility of the infrastructure in forested areas. We did not find a significant effect of traffic intensity on the MSA of birds. Species varied in their response to infrastructure. Raptors were found to be more abundant in the proximity of infrastructure whereas other bird taxa tended to avoid it. Abundances were affected at variable distances from infrastructure: within a few meters for small-sized mammals and up to several hundred meters for large-sized mammals.

Our findings show the importance of minimizing infrastructure development for wildlife conservation in relatively undisturbed areas. By combining actual species distributions with the effect distance functions we developed, regions sensitive to infrastructure development may be identified. Additionally, the effect distance functions can be used in models in support of decision making on infrastructure planning.

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## 1. Introduction

Global biodiversity is changing at an unprecedented rate as a result of several human-induced changes in the global environment (Vitousek, 1994; Pimm et al., 1995; Sala et al., 2000; MEA, 2005). Biodiversity loss at the species level tends to result in the so-called homogenisation process (Lockwood and McKinney, 2001). This process is generally characterised by a decrease in the abundance of many species, culminating into an increase in the number of threatened species and the extinction of others, in combination with a simultaneous increase in the abundance of a few species.

The main drivers of biodiversity change are land-use and land-cover change, climate change, pollution, fragmentation and infrastructure development (UNEP, 2001; Sala et al., 2000; Sanderson et al., 2002; Alkemade et al., 2009).

The ubiquity of road networks and the growing body of evidence of the negative impacts that roads and other linear infrastructure have on wildlife and ecosystems suggest that infrastructure represents a major driving factor of biodiversity loss. The most commonly reported impacts from roads and utility corridors include habitat loss, intrusion of edge effects in natural areas, isolation of populations, barrier effects, road mortality and increased human access (Andrews, 1990; Forman and Alexander, 1998; Spellerberg, 1998; Trombulak and Frissell, 2000; Forman et al., 2003). Road construction leads to habitat destruction and creates open spaces in otherwise closed forests (Gullison and Hardner, 1993; Reed et al., 1996; Santos and Tabarelli, 2002). The open spaces may fragment populations (barrier effect), attract light-demanding species and may be avoided by others (edge effect)

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(Kroodsmma, 1984; Vos and Chardon, 1998; Bolger et al., 1997; Ortega and Capen, 1999). Additionally, the use of infrastructure by cars or trains increases the risk of collisions with wildlife and the stress on (breeding) individuals (due to noise and visual stimuli), both of these risks affecting animal populations (Van der Zande et al., 1980; Reijnen et al., 1996; Romin and Bissonette, 1996; Boorman and Sazaki, 2005; Parris and Schneider, 2009).

Besides roads, other types of infrastructure, such as railways, powerlines, pipelines, hydroelectric developments, oil wells, seismic lines and wind parks, have an impact on wildlife populations (Dunthorn and Errington, 1964; McLellan and Shackleton, 1989; Cameron et al., 1992; Van Dyke and Klein, 1996; Mahoney and Schaefer, 2002; Nellemann et al., 2003a; Barrios and Rodriguez, 2004). All these impacts may influence the long-term viability of populations and, eventually, biodiversity.

Qualitative reviews provide a broad understanding of the ecological effects of infrastructure that affect a range of taxa and ecosystems, but lack quantitative evidence (Trombulak and Frissell, 2000; Forman et al., 2003). However, the few attempts to quantify the effects of infrastructure (UNEP, 2001; Nellemann et al., 2003b; Fahrig and Rytwinski, 2009), or to model the vulnerability of animal populations to road effects (Jaeger et al., 2005), are not based on meta-analysis, which is the statistical procedure for combining the results of independent studies in a quantitative way (Arnqvist and Wooster, 1995). In this study, we aim at estimating the decline of animal populations in relation to proximity to infrastructure by using a meta-analytical approach.

Among all animal taxa, mammal and bird populations were chosen for our analysis since both have been widely reported to be declining in relation to their distance from infrastructure. However, large differences in disturbance sensitivity seem to exist between and within these groups. Bird populations seem to be affected within a few hundred metres from infrastructure, whereas a reduction in mammal populations has been found at distances of a few hundred metres up to several kilometres from infrastructure (McLellan and Shackleton, 1989; Cameron et al., 1992; Ortega and Capen, 1999; Nellemann et al., 2003a). Additionally, traffic intensity seems to play a role in the decline of both bird and mammal populations close to roads (Van der Zande et al., 1980; Reijnen et al., 1995, 1996; Dyer et al., 2001; Rheindt, 2003; Gagnon et al., 2007).

To quantify the patterns of reduced population densities in relation to infrastructural development, we searched the scientific and non-scientific literature for quantitative data on mammal and bird populations at varying distances from infrastructure. As the metric of effect size, we calculated the ratio between the species abundance at varying distances to infrastructure (Disturbance or Effect distance) relative to the species abundance at the largest (control) distance reported in the study. This ratio is a form of the biodiversity indicator mean species abundance (MSA) which represents the mean abundance of (remaining) original species in an area related to an undisturbed situation (Alkemade et al., 2009). Meta-analysis was used to combine the effect sizes (MSA values) across all studies for several distance intervals and test their level of significance. Furthermore, meta-regression was applied to model the relationship between distance to infrastructure and MSA for birds ( $MSA_B$ ) and mammals ( $MSA_M$ ) (infrastructure–distance effect), and to examine sources of heterogeneity in this relationship.

## 2. Methods

### 2.1. Search and selection of published studies on infrastructural effects

Relevant studies were searched by using the following electronic databases: Ebsco, ISI Web of Knowledge, JSTOR, Omega (Utrecht University Digital Publications Search Machine), Science Direct, Scopus,

Springer Link and Wiley InterScience. The search terms were: road\* AND impact\* AND biodiversity OR mammal, bird; infrastructure AND impact\* AND biodiversity OR mammal, bird; road\* AND distance AND biodiversity OR mammal, bird; road-effect zone AND mammal abundance, bird abundance; road\* AND disturbance\* AND biodiversity OR mammal, bird; powerline AND impact AND biodiversity OR mammal, bird; wind park AND biodiversity OR mammal, bird; road traffic\* AND impact\* AND biodiversity\* OR mammal, bird; infrastructure AND disturbance AND biodiversity OR mammal, bird. An Internet search was also performed using the meta-search engine Google scholar. Bibliographies of articles viewed at full text were searched for relevant secondary articles. Authors and recognized experts in the field of infrastructure development, road establishment and effects on biodiversity (Christian Nellemann, UNEP-Grid Arendal, and Rien Reijnen, Alterra) were also contacted for further recommendations, and for provision of any unpublished material or missing data that may be relevant (grey literature). Foreign language searches were undertaken by using cross-reference.

### 2.2. Study inclusion criteria

From this bulk of literature we selected those studies of which title and keywords were associated to the objective of this review. Subsequently, information contained in the abstracts was examined to further narrow down the selection to those studies that met the following criteria:

- Relevant study objects: Populations of any mammal or bird species. Studies were included irrespective of habitat or spatial scale.
- Types of intervention: Disturbance distances or distances close to infrastructure at which mammal and bird populations might be reduced compared to larger distances or control distances (see Types of comparator).
- Types of outcome: Species abundance (density and/or counts) at varying distances to infrastructure.
- Types of comparator: Control distances or distances at which mammal and bird populations are unaffected by infrastructure and roads.

### 2.3. Data extraction

Finally, 49 studies met the selection criteria for data extraction, from which 90 datasets were extracted and stored in a database, resulting in 2107 data points. The data included the mean abundance at disturbance distances close to infrastructure and at a larger control distance; furthermore we recorded the sample size, the variance, and standard deviation or standard error, depending on the study. These data were used to estimate an effect size and its variance as required in meta-analysis (Osenberg et al., 1999). Additionally, we stored data on location, habitat, infrastructure type, taxon (order) and traffic intensity to explore sources of heterogeneity (see Table 2 in Supplementary material, available at <http://www.environmentalevidence.org/SR68.html>). These variables are considered biologically meaningful and could affect the way different taxa respond to infrastructure. Thus, we expected that different taxa would respond differently to different infrastructure types (linear and clustered) and in different habitat types according to varying visibility of infrastructure, while traffic intensity could affect the response due to the influence of noise and visual stimuli.

### 2.4. Effect size calculation: Mean Species Abundance (MSA)

For each study, individual effect sizes were calculated as the ratio between the abundance of each species close to the infrastructure

(Disturbance distance) and the abundance of the same species at the largest (control) distance, as reported in the study. Individual effect sizes were aggregated for each study and distance, resulting in an estimate of the mean species abundance (MSA), which is the metric of effect size for the meta-analysis (see Eq. (1))

$$MSA_{sd} = \frac{\sum_i R_{isd}}{N_s}; \quad (1)$$

where  $MSA_{sd}$  is the relative mean species abundance estimated in study  $s$  at a distance  $d$ ;  $R_{isd}$  is the ratio between the abundance or density of species  $i$  at distance  $d$  and the abundance or density of species  $i$  at the control distance, calculated as:  $A_{isd}/A_{isc}$  for  $A_{isc} > 0$ .  $N_s$  is the number of species considered in study  $s$ . MSA values ranged from 0 to 1 and declined at shorter distances from infrastructure. For species with higher densities at short distances from infrastructure compared to the control distance, the MSA value was truncated to 1; therefore, if  $A_{isd} > A_{isc}$ , then  $R_{isd} = 1$ .

### 2.5. Estimation of variation in MSA values

The variance of the MSA value for each distance and study was estimated by calculating the variance of the external error (2), and of the internal error (3), which are both forms of the variance of a sample mean (Mood et al., 1973). The largest variance was used in the meta-analysis, thus taking into account the largest error associated with each data point (DerSimonian and Laird, 1986). For single species' studies, only the variance of the internal error could be calculated.

The variance of the external error was calculated as:

$$\sigma_{ext}^2 = \frac{\sum (MSA_{sd} - R_{isd})^2}{N_s(N_s - 1)} \quad (2)$$

The variance of the internal error was calculated as:

$$\sigma_{int}^2 = \frac{\sum \sigma_{R_{isd}}^2}{N_s^2}; \quad (3)$$

where  $\sigma_{R_{isd}}^2$  is the individual variance for each ratio, which was calculated by using the Delta Method (4), a first-order approximation of the variance of a ratio of two random variables (Oehlert, 1992; Winzer, 2000).

$$\sigma_{R_{isd}}^2 = \frac{A_{isd}^2}{A_{isc}^2} \left[ \frac{A_{isd}^2}{\sigma_{A_{isd}}^2} + \frac{A_{isc}^2}{\sigma_{A_{isc}}^2} - \frac{2\rho \cdot \sigma_{A_{isd}} \cdot \sigma_{A_{isc}}}{A_{isd} \cdot A_{isc}} \right] \quad (4)$$

In this equation  $\sigma_{A_{isd}}^2$  and  $\sigma_{A_{isc}}^2$  are the variances of  $A_{isd}$  and  $A_{isc}$ , respectively, and  $\rho$  their correlation coefficient. We assume  $A_{isd}$  and  $A_{isc}$  to be independent and, therefore, the correlation coefficient  $\rho$  to be zero. Variances of  $A_{isd}$  and  $A_{isc}$  were obtained from studies, when available; where this was not the case, the data was assumed to follow a Poisson distribution, in which  $\mu = \sigma^2$  and, therefore,  $A_{isd} = \sigma_{A_{isd}}^2$ , and  $A_{isc} = \sigma_{A_{isc}}^2$  (Sokal and Rohlf, 1981).

Finally, as some species had a density of zero at the disturbance distance ( $A_{isd} = 0$ ), a continuity correction factor ( $k = 1/2$ ) was added to the numerator and denominator of the ratio of all species, resulting in slightly higher variance estimates (Cox, 1970; Sweeting et al., 2004).

### 2.6. Study quality assessment

Study characteristics were summarized and experimental design (control and treatment plots) and data availability for extraction (means, standard errors and sample sizes) were used as criteria for determining study quality (low; medium-low; medium; medium-high; high) (Supplementary material available at <http://www.environmentalevidence.org/SR68.html>). A sensitivity

analysis was done by removing studies scoring "medium-low" or "low".

### 2.7. Data analyses

Meta-analyses were performed separately for mammal and bird studies by using the package "metafor" in R 2.9.1 software (Viechtbauer, 2009). A random effects meta-analysis was done to derive a pooled effect size for all datasets allowing pseudoreplication. Additionally, meta-analyses were done per distance interval containing non-duplicated independent datasets.

Heterogeneity was assessed by inspection of Forest plots and formal tests of heterogeneity  $Q$  and  $I^2$  (Thompson and Sharp, 1999). Publication bias was also assessed using Funnel plots of asymmetry along with formal tests (Egger et al., 1997; Supplementary material available at <http://www.environmentalevidence.org/SR68.html>).

To explore factors introducing heterogeneity we built several Generalized Linear Mixed Models (GLMM), accounting for several alternative nested ecological hypotheses that included the following a priori selected explanatory variables: distance to infrastructure (DIST or LOGDIST when log-transformed), presence of forest cover (FOR), infrastructure type (INFYTP) and traffic intensity (TRAF). All GLMM were built in S-Plus 7.0 and fit by restricted penalised quasi-likelihood (Pinheiro and Bates, 2000). Each MSA value was weighed by its variance. Study was introduced as random effect since we expected similar but not identical effects of infrastructure across studies.

Models were compared and selected by means of information theoretic criteria, including Akaike's Information Criterion corrected for sample size (AICc) and Akaike weights. AIC corrected for overdispersion (QAIC) was not needed since the random effects of the GLMM accommodate any possible overdispersion in the data. This was *ad hoc* checked by calculating a scale parameter (sigma) for our models using package "lme4" in R 2.9.1. (Bates and Maechler, 2009). The model selected was that minimizing the loss of Kullback–Leibler information.

Additionally data was disaggregated and GLMM were built to examine differences in the relationship between MSA and distance for different habitats, for forested and non-forested habitats, for different infrastructure types (linear and clustered) and for different taxa.

## 3. Results

### 3.1. Data availability and selected studies: review statistics

More than 600 studies contained relevant titles and abstracts. Of these, 50 studies corresponded to the selected criteria for data extraction. Two studies referred to the same data and were treated as one (Noel et al., 2004; Joly et al., 2006), resulting in 49 studies used for the meta-analysis. Some geographical bias was found since most of the studies were from either North America (21) or Europe (23), while a few studies from Africa (3) and Oceania (2) were found.

Twenty-seven studies included 201 bird species, and 49 independent datasets were extracted for the meta-analysis. The other 22 studies included 33 mammal species, and 41 independent datasets were extracted. Some species were repeated more than once (Appendix 3, Supplementary material available at <http://www.environmentalevidence.org/SR68.html>). Of the 49 datasets for birds, 10 contained relevant information on traffic intensities. Of the 41 datasets for mammals, five included information on traffic intensity, which was considered insufficient for the inclusion of this variable in the analysis (Supplementary material available at <http://www.environmentalevidence.org/SR68.html>).

Bird datasets frequently included a large number of species (mean = 9.1 (1–54)), compared to mammal datasets, which (with some exceptions, e.g. Newmark et al., 1996; Goosem and Marsh, 1997; Yost and Wright, 2001) usually focused on a single species (mean = 3.7 (1–11)). The most represented habitat types within the bird datasets were grasslands and agricultural lands (each of them in 15 datasets), and the least represented was boreal forests (1 dataset). The most represented habitat type in the mammal datasets was arctic tundra (12 times) and the least represented habitat types were grasslands and semi-arid habitats (1 time each).

The most represented bird taxon was Passeriformes (21 datasets) and the least represented bird taxa were Coraciiformes, Psittaciformes and Trochiliformes (1 dataset each). The most represented mammal taxon was Artiodactyla (25 datasets) and the least represented mammal taxa were Lagomorpha and Perissodactyla (1 dataset each).

Reported distances in bird datasets were in the range of 0–2580 m whereas data points for mammals were obtained within a range of 0–17,000 m.

3.2. Meta-analysis

3.2.1. Combination of all distances with pseudoreplication

The pooled effect size derived from an all-encompassing meta-analysis of MSA values for birds indicated that bird abundance declined within ca. 2600 m from infrastructure ( $MSA_B = 0.678$ ; 95% CI 0.636–0.720,  $P < 0.0001$ ; Table 1). Likewise for mammals,  $MSA_M$  decreased within 17,000 m from infrastructure ( $MSA_M = 0.675$ ; 95% CI 0.608–0.742,  $P < 0.0001$ ). However, for both meta-analyses

there was large heterogeneity and publication bias within the datasets ( $Q_B = 16938.28$ ,  $P < 0.0001$ ; Egger test<sub>B</sub> = 5.785,  $P < 0.0001$ ;  $Q_M = 3466.80$ ,  $P < 0.0001$ ; Egger test<sub>M</sub> = 3.684,  $P < 0.0001$ ; Tables 1 and 2). Fail-safe numbers indicated that a large number of studies reporting neutral or positive effects of the proximity of infrastructure on species abundance would be needed to overturn these results and so even with some publication bias, the results for mammals and birds can be considered a reliable estimate of the true effect (Rosenthal, 1979).

Sensitivity analyses were performed and the results remained similar after removing studies that scored “medium–low” in the study quality assessment. The pooled effect size for bird data increased slightly, heterogeneity was lower but still statistically significant and there was evidence of publication bias ( $MSA_B = 0.683$ ; 95% CI 0.627–0.740,  $P < 0.0001$ ;  $Q = 2653.70$ ,  $P < 0.0001$ ; Egger test = 4.699,  $P < 0.0001$ ; Appendix 4, Supplementary material available at <http://www.environmentalevidence.org/SR68.html>). For mammals the pooled effect size was similar, heterogeneity decreased slightly and there was publication bias ( $MSA_M = 0.678$ ; 95% CI 0.6086–0.7472,  $P < 0.0001$ ;  $Q = 3401.70$ ,  $P < 0.0001$ ; Egger test = 4.006,  $P < 0.0001$ ; Appendix 4, Supplementary material available at <http://www.environmentalevidence.org/SR68.html>).

3.2.2. Effect size per distance interval

Pooled effect sizes calculated per distance interval for independent datasets were significant for mammal and bird data, but considerable heterogeneity and publication bias existed for most of the intervals. Lower MSA values were obtained at shorter distance intervals to infrastructure for both mammals and birds.

**Table 1**  
Results of the meta-analysis for bird species at different distance intervals.

Distance (m)	n	Effect size (MSA)	SE	CI (lb)	CI (ub)	P (e.size)	Q	P(Q)	I <sup>2</sup> (%)	Egger's test intercept	P(t) Egger	Fail-safe N
0–2580	288 <sup>a</sup>	0.6777	0.0215	0.6355	0.7200	<0.0001	16938.28	<0.0001	96.6	5.785	<0.0001	11,23,452
<10	10	0.3983	0.1290	0.1455	0.6512	0.002	62.7322	<0.0001	94.4	2.625	0.0026	245
15–35	18	0.4855	0.0893	0.3105	0.6605	<0.0001	223.89	<0.0001	93.3	4.336	<0.0001	2233
38–65	20	0.5339	0.0905	0.3566	0.7112	<0.0001	370.28	<0.0001	95.2	3.639	<0.0001	1939
70–80	16	0.5923	0.0896	0.4165	0.7680	<0.0001	33.19	0.007	54.5	2.561	0.0002	689
90–100	16	0.6218	0.0722	0.4802	0.7634	<0.0001	38.42	0.0004	68.4	3.993	0.0003	1494
110–125	13	0.6673	0.1022	0.4671	0.8676	<0.0001	40.99	<0.0001	65.7	3.072	0.0001	577
130–140	5	0.7070	0.1592	0.3950	1.0190	<0.0001	45.79	<0.0001	85	4.981	0.0222	225
150–160	16	0.5978	0.0788	0.4434	0.7522	<0.0001	79.12	<0.0001	83.8	4.874	0.0002	1946
170–180	10	Fisher scoring algorithm did not converge										
190–200	13	0.6292	0.0908	0.4512	0.8072	<0.0001	301.50	<0.0001	95.7	6.786	0.0015	2864
210–240	12	0.6734	0.1281	0.4223	0.9246	<0.0001	428.83	<0.0001	95.3	5.434	0.0124	1560
250–280	17	0.6676	0.0963	0.4789	0.8563	<0.0001	331.47	<0.0001	96.5	9.347	0.0122	9320
300–320	15	0.7454	0.1062	0.5374	0.9535	<0.0001	8769.78	<0.0001	99.4	15.540	0.202	20,053
340–375	10	0.6432	0.0943	0.4583	0.8281	<0.0001	34.74	0.0001	75.2	4.828	0.0161	851
380–480	16	0.7495	0.0888	0.5755	0.9236	<0.0001	1052.78	<0.0001	96.9	8.952	0.0003	7820
490–550	14	0.6946	0.1004	0.4978	0.8914	<0.0001	86.19	<0.0001	87	5.331	0.0073	2013
565–645	9	0.7182	0.0929	0.5361	0.9003	<0.0001	27.72	0.0005	75.8	6.594	0.0436	1294
650–785	11	0.7564	0.1203	0.5206	0.9921	<0.0001	581.18	<0.0001	94.1	5.980	0.0451	1588
800–860	3	0.6869	0.1968	0.3011	1.0728	<0.0001	12.75	<0.0001	81.1	4.640	0.0124	69
900–915	4	0.9152	0.1043	0.7108	1.1196	<0.0001	0.2687	0.9658	0	4.1168	0.0189	97
1000–1075	11	0.8363	0.0791	0.6812	0.9913	<0.0001	17.23	0.0695	46.7	4.585	0.0050	930
1100–1175	3	0.9696	0.1275	0.7198	1.2195	<0.0001	0.0722	0.9645	0	4.052	0.0780	52
1200–1290	4	0.8308	0.1097	0.6158	1.0459	<0.0001	5.011	0.171	47	5.426	0.0344	171
1300	1	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
1400	2	0.9098	0.0515	0.8090	1.0107	<0.0001	0.0398	0.842	0	10.941	0.321	176
1500–1505	7	0.8511	0.0677	0.7183	0.9838	<0.0001	13.894	0.0308	71.7	10.572	0.0516	2018
1600	1	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
1700–1750	4	0.9294	0.0992	0.7349	1.1239	<0.0001	0.5252	0.9133	0	4.4439	0.0148	113
1800–2000	4	1.0000	0.0483	0.9053	1.0947	0.0000	0	1.0000	0	7.868	0.136	362
2150	1	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
2365	1	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
2580	1	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.

<sup>a</sup> Total number of MSA values included in the meta-analysis for bird species. The total number of studies is 27, from which 49 datasets were extracted containing 288 MSA calculated values for different distances.

**Table 2**  
Results of the meta-analysis for mammal species at different distance intervals.

Distance (m)	n	Effect size (MSA)	SE	CI (lb)	CI (ub)	P (e.size)	Q	P(Q)	I <sup>2</sup> (%)	Egger's test intercept	P(t) Egger	Fail-safe N
0–17,000	151 <sup>a</sup>	0.6746	0.0342	0.6076	0.7415	<0.0001	3466.80	<0.0001	90.7	3.6843	<0.0001	114,151
1	6	0.1528	0.1005	-0.0442	0.3498	>0.1	12.74	0.0259	71.9	1.0862	0.0937	10
10–25	11	0.7110	0.0451	0.6227	0.7993	<0.0001	4.38	0.9284	0	4.300	<0.0001	816
30–50	16	0.5651	0.0726	0.4229	0.7073	<0.0001	28.65	0.0178	55.8	2.9972	<0.0001	834
75–100	7	0.3957	0.1831	0.0369	0.7545	<0.05	10.87	0.0925	47.7	1.4193	0.0016	362
110–180	16	0.8374	0.0520	0.7354	0.9394	<0.0001	7.42	0.9448	0	3.4069	<0.0001	1083
200	2	0.6104	0.2382	0.1435	1.0774	0.0104	0.02	0.878	0	1.464	0.402	2
250–300	9	0.8470	0.0627	0.7241	0.9698	<0.0001	7.78	0.4557	7.8	4.059	0.0037	485
350–600	19	0.6222	0.1115	0.4035	0.8408	<0.0001	206.68	<0.0001	91.9	3.3561	<0.0001	1485
750–1000	6	0.8669	0.1052	0.6608	1.0731	<0.0001	9.23	0.1002	50.2	4.8430	0.0036	307
1050–2200	20	0.5786	0.0806	0.4207	0.7366	<0.0001	75.49	<0.0001	74.1	3.0049	<0.0001	1316
2500	8	0.8233	0.2098	0.4121	1.2345	<0.0001	0.9453	0.9957	0	1.2516	0.0020	30
3500–4000	7	0.9807	0.1276	0.7307	1.2308	<0.0001	0.2907	0.9995	0	1.9095	0.0775	60
4500–5000	8	0.8666	0.1099	0.6512	1.0820	<0.0001	6.06	0.5323	30.8	3.395	0.0255	265
5500–7000	8	0.8049	0.1983	0.4163	1.1936	<0.0001	0.8083	0.9974	0	1.2467	0.0039	29
7500	2	0.8730	0.0118	0.8498	0.8962	<0.0001	0.0084	0.9272	0	37.35	0.494	2047
8500	1	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
9500–11,000	4	1.0000	0.0131	0.9744	1.0256	<0.0001	0	1	0	76.122	<0.0001	2527
17,000	1	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.

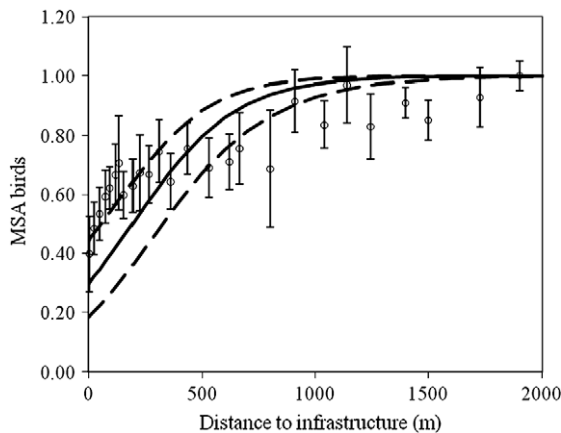
<sup>a</sup> Total number of observations points or MSA values included in the meta-analysis for mammal species. The total number of studies is 22, from which 41 datasets were extracted containing 151 MSA calculated values for different distances.

The number of datasets per distance interval decreased as the distance from infrastructure increased (Tables 1 and 2).

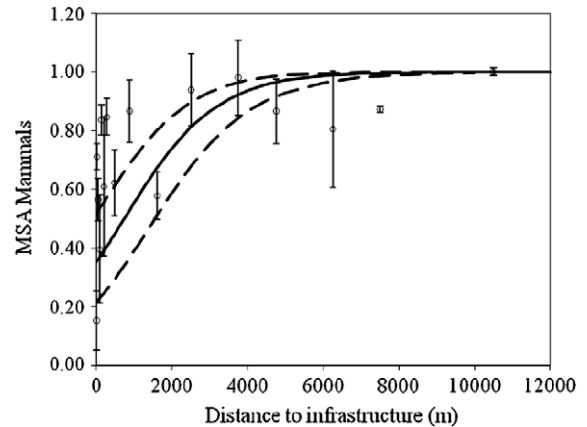
Sensitivity analyses for mammal and bird data resulted in similar MSA values per distance interval with the exception of the distance intervals 300–320, 340–375 and 380–490 m for the bird data, which had larger MSA values (Appendix 4, Supplementary material available at <http://www.environmentalevidence.org/SR68.html>). Heterogeneity was not statistically significant in these intervals, but there was publication bias.

3.2.3. Exploration of reasons for heterogeneity: meta-regression

The relationship between MSA and distance to infrastructure was positive for both mammals and birds (Fig. 1 and 2). When the data was subgrouped per habitat, all relationships were also positive except for temperate forests in the case of mammal species, and Mediterranean forests in the case of bird species (Tables 6 and 7). In forests both bird and mammal species abundances were affected in the proximity of infrastructure whereas in non-forested areas the effect extended over a larger distance (Tables



**Fig. 1.** Logistic regression between mean species abundance of birds and distance from infrastructure. Open dots represent the pooled results of the meta-analysis per distance interval ± S.E. The black line denotes the estimated curve for the decline of MSA, related to distance. Dashed lines are the 95% upper and lower limits of the confidence bands of the curve.



**Fig. 2.** Logistic regression between mean species abundance of mammals and distance from infrastructure. Open dots represent the pooled results of the meta-analysis per distance interval ± S.E. The black line denotes the estimated curve for the decline of MSA, related to distance. Dashed lines are the 95% upper and lower limits of the confidence bands of the curve.

6 and 7). All relationships had lower AICc when “LOGDIST” was chosen as explanatory variable, except for Tundra.

There existed also differences in the relationship between MSA and distance to infrastructure for different taxa. Accitriformes and Falconiformes were the only bird taxa which were unaffected or positively affected by the presence of infrastructure, whereas for other bird taxa proximity to infrastructure seemed to exert a negative effect on species abundance, or the magnitude of the effect was unknown due to low sample size (Supplementary material available at <http://www.environmentalevidence.org/SR68.html>). From all mammal taxa, a positive relationship between MSA and distance to infrastructure could only be found for Artiodactyla and Rodentia, but abundances of the latter were only reduced at short distances from infrastructure while this effect extended over a large distance for the former.

Several *ad hoc* models were built to explore the high heterogeneity between effect sizes. We worked on several biological hypotheses to explain the variability in the relationship between MSA and distance to infrastructure. For bird species the most parsimonious model was that containing only the explanatory variable “LOGDIST” (Akaike weight: 0.78, Table 3). For mammal

**Table 3**  
Models expressing different biological hypotheses on the influence of infrastructure distance, infrastructure type and infrastructure visibility (forest) on the mean species abundance of birds. The selected model appears in bold.

Model (birds)	AICc	K	n	$\Delta_i$ AICc	$w_i$
LOGDIST + FOR + INFITYP + LOGDIST * INFITYP + LOGDIST * FOR + FOR * INFITYP + LOGDIST * INFITYP * FOR	No model convergence				
LOGDIST + FOR + LOGDIST * FOR	1630.78	6	288	3.04	0.17
LOGDIST + INFITYP + LOGDIST * INFITYP	1636.04	6	288	8.30	0.01
LOGDIST + FOR	1633.96	5	288	6.21	0.03
LOGDIST + INFITYP	1637.29	5	288	9.54	0.01
LOGDIST	<b>1627.75</b>	<b>4</b>	<b>288</b>	<b>0.00</b>	<b>0.78</b>
FOR	1647.18	4	288	19.43	<0.01
INFITYP	1646.40	4	288	18.65	<0.01
DIST + FOR + DIST * FOR	1762.19	6	288	134.45	<0.01
DIST + FOR	1744.61	5	288	116.86	<0.01
DIST	1739.98	4	288	112.23	<0.01

AICc = Akaike Information Criterion corrected for sample size;

K = No. of estimable parameters: Intercept, log (effect distance), effect distance, forest, infra type, interaction term, study and error term;

n = No. of data points;

$\Delta_i$ AICc = Delta AICc value;

$w_i$  = Akaike weight.

**Table 4**  
Models expressing different biological hypotheses on the influence of infrastructure distance, infrastructure type and infrastructure visibility (forest) on the mean species abundance of mammals. The selected model appears in bold. See Table 3 for explanation of coefficients.

Model (mammals)	AICc	K	n	$\Delta_i$ AICc	$w_i$
LOGDIST + FOR + INFITYP + LOGDIST * INFITYP + LOGDIST * FOR + FOR * INFITYP + LOGDIST * INFITYP * FOR	889.18	10	151	3.59	0.14
LOGDIST + INFITYP + LOGDIST * INFITYP	963.06	6	151	77.50	<0.01
LOGDIST + FOR + LOGDIST * FOR	900.02	6	151	14.46	<0.01
LOGDIST + LOGDIST	919.67	5	151	34.12	<0.01
LOGDIST + FOR	911.34	5	151	25.79	<0.01
LOGDIST	923.42	4	151	37.87	<0.01
INFITYP	896.93	4	151	11.38	<0.01
FOR	899.30	4	151	13.75	<0.01
DIST + FOR + INFITYP + DIST * INFITYP + DIST * FOR + FOR * INFITYP + DIST * INFITYP * FOR	1081.87	10	151	196.28	<0.01
DIST + FOR + DIST * FOR	964.56	6	151	79.01	<0.01
DIST + INFITYP	893.28	5	151	7.73	0.02
DIST + FOR	<b>885.54</b>	<b>5</b>	<b>151</b>	<b>0.00</b>	<b>0.84</b>
DIST	897.99	4	151	12.45	<0.01

**Table 5**  
Models expressing different biological hypotheses on the influence of infrastructure distance, traffic intensity (noise) and infrastructure visibility (forest) on the mean species abundance of birds. The selected model appears in bold. See Table 3 for explanation of coefficients.

Model (birds)	AICc	K	n	$\Delta_i$ AICc	$w_i$
LOGDIST + FOR + TRAF + LOGDIST * TRAF + LOGDIST * FOR + FOR * TRAF + LOGDIST * TRAF * FOR	583.52	10	87	10.87	<0.01
LOGDIST + TRAF + LOGDIST * TRAF	512.30	6	87	31.65	<0.01
LOGDIST + TRAF	503.06	5	87	22.40	<0.01
TRAF	537.10	4	87	56.44	<0.01
LOGDIST	<b>480.66</b>	<b>4</b>	<b>87</b>	<b>0.00</b>	<b>0.99</b>

species the most parsimonious model was that including the explanatory variables “DIST” and “FOR” (Akaike weight: 0.84, Table 4).

The most parsimonious model for the subset of bird data containing information on traffic intensity was that including only “LOGDIST” as explanatory variable (Akaike weight: 0.99, Table 5).

## 4. Discussion

### 4.1. Effect of distance from infrastructure on bird and mammal species abundance

Our analyses suggest that infrastructure can have a negative impact on bird and mammal abundance and that this impact is more evident in the proximity of the infrastructure. Pooled results for pseudoreplicated data indicated a decline in species abundance

of 28–36% and 25–38% for birds and mammals within 2.6 km and 17 km from infrastructure, respectively. Pooled results per distance interval showed that MSA of birds and of mammals became higher for distance intervals far away from infrastructure. The meta-regression also indicated a positive relationship between MSA of birds and mammals and distance to infrastructure. A second conclusion that can be obtained from these results is that bird populations are likely to be more affected at short distances from infrastructure while the effect on mammal populations seems to extend over larger distances. These results confirm the effect distances reported in other studies, which were larger for mammals (Cameron et al., 1992; Newmark et al., 1996; Nellemann et al., 2003a; Joly et al., 2006) than for birds (Van der Zande et al., 1980; Madsen, 1985; Reijnen et al., 1996; Rheindt, 2003). However, considerable heterogeneity was found in our results, especially for birds, and also publication bias, both limiting the robustness of these conclusions.

**Table 6**

Univariate meta-regression coefficients for the relationship between MSA and distance for bird species in different habitats. Models with log-transformed distance as explanatory variable are more parsimonious than without transformation for all habitats.

Bird species							
Explanatory variable	Habitat	<i>b</i>	Intercept	<i>k</i>	<i>n</i>	<i>K</i>	AICc
LOGDIST	Agricultural lands	1.523	−7.933	15	109	4	626.29
	Temperate forests	0.761	−2.868	8	35	4	192.57
	Boreal forests	No model convergence		1	2		
	Mediterranean forests	27.020	−136.950	2	4	4	−12.28
	Grasslands	1.014	−5.193	15	97	4	417.24
	Scrublands	1.174	−4.853	5	35	4	200.84
	Tropical forests	18.834	−61.705	3	6	4	80.13
	All forests	0.826	−2.770	14	47	4	263.04
	Non-forested habitats	1.333	−6.712	35	241	4	1361.75
DIST	Agricultural lands	0.005	−1.777	15	109	4	779.75
	Temperate forests	No model convergence		8	35	4	
	Boreal forests	No model convergence		1	2		
	Mediterranean forests	0.046	3.222	2	4	4	0.45
	Grasslands	0.002	−0.759	15	97	4	488.01
	Scrublands	No model convergence		5	35	4	
	Tropical forests	0.522	−17.254	3	6	4	97.57
	Forests	0.011	−0.894	14	47	4	301.93
	Non-forested habitats	0.004	−1.218	35	241	4	1478.68

**Table 7**

Univariate meta-regression coefficients for the relationship between MSA and distance for mammal species in different habitats. Models with log-transformed distance as explanatory variable are more parsimonious than without transformation for all habitats except for tundra.

Mammal species							
Explanatory variable	Habitat	<i>b</i>	Intercept	<i>k</i>	<i>n</i>	<i>K</i>	AICc
LOGDIST	Agricultural lands	No model convergence		2	9		
	Temperate forests	0.260	0.103	4	15	4	67.99
	Boreal forests	1.526	−7.742	9	34	4	162.23
	Grasslands	Only 1 dataset		1	5		
	Scrublands	2.133	−6.773	3	11	4	59.19
	Tropical forests	0.681	−1.136	5	22	4	81.39
	Semi-arid lands	Only 1 dataset		1	8		
	Arctic tundra	2.854	−21.429	12	52	4	364.50
	All except for Tundra	0.978	−3.670	25	104	4	528.05
	Forests	0.860	−3.151	18	71	4	313.90
	Non-forested habitats	1.142	−7.089	19	85	4	548.28
	DIST	Agricultural lands	No model convergence		2	9	
Temperate forests		0.002	0.850	4	15	4	75.91
Boreal forests		No model convergence					
Grasslands		Only 1 dataset		1	5		
Scrublands		No model convergence		4			
Tropical forests		0.003	0.592	5	22	4	107.21
Semi-arid lands		Only 1 dataset		1	8		
Arctic tundra		0.001	−1.832	12	52	4	332.68
All except for Tundra		0.004	−0.478	25	104	4	614.09
Forests		0.003	−0.026	18	71	4	397.21
Non-forested habitats		0.001	−1.659	19	85	4	561.76

The sensitivity analyses resulted in a slightly (but not statistically significant) larger pooled effect size for pseudoreplicated data in the case of bird species. Sensitivity analyses per distance interval showed similar results to the full meta-analyses except for the distance intervals between 300 and 480 m from infrastructure, with larger effect sizes. Yet, fail-safe numbers indicate that a large number of non-significant studies would be needed to overturn the pooled effect sizes calculated for these distance intervals. Thus, we decided to maintain these studies in the meta-regression following Wolf and Guevara (2001), who advocate for the use of all available data when doing meta-analysis.

The high heterogeneity underlying the results of our meta-analysis indicates that infrastructure development can have negative impacts on bird and mammal abundance within a certain distance depending on a number of factors which we further explore in Section 4.2.

#### 4.2. Exploration of sources of heterogeneity

Meta-regression helped us to elucidate that “LOGDIST” was the main explanatory variable for the decline in abundance of bird populations due to infrastructure. For mammals, “DIST” was the main variable but the variable “FOR” seemed to add important information to the model. Meta-regression on subgroups divided by habitat type and taxa indicated that in open habitats, both mammal and bird populations seem to avoid infrastructure over larger distances, compared to those in forested biomes, which could probably be related to reduced visibility of infrastructure. Forman and Deblinger (2000) showed similar results for breeding birds in open grasslands and in woodlands (data adapted from Reijnen et al. (1995, 1996)).

For the subset of bird data including information on traffic intensity, the most parsimonious model was that which contained

only “LOGDIST” as explanatory variable. Therefore it seems that traffic intensity has no effect on the reduction of bird populations nearby roads. These results are contrary to the findings of a number of authors that have highlighted the pernicious effects of traffic intensity and noise on bird populations (Reijnen and Foppen, 1994; Reijnen et al., 1995, 1996, 1997; Forman et al., 2002; Rheindt, 2003). However, there are other studies which found a decline in bird populations near roads with low traffic intensity (Räty, 1979; Madsen, 1985; Develey and Stouffer, 2001) and others which found no clear relationship (Peris and Pescador, 2004). Finally, some authors claim that there exists a trade-off between traffic intensity and velocity, with low traffic intensity being related to higher velocities (Martínez-Abraín, 1994; Drews, 1995). Yet, traffic velocity seems to be related to bird mortality, which occurs on the road itself. To the best of our knowledge, studies that deal with this topic usually do not report on bird densities at increasing distances from roads (or include a control distance for comparison). As none of the studies included in our analysis contained data on traffic velocity, the influence of this variable could not be evaluated.

Not all species showed a decline in abundance nearby infrastructure: species abundance of Accipitriformes and Falconiformes was larger in the proximity of infrastructure. This was not a surprising outcome since other studies have reported the presence of raptors nearby roads searching roadkill carrion (Forman and Alexander, 1998; Lambertucci et al., 2009) and hunting (Dónazar et al., 1993; Fajardo et al., 1998; Dean and Milton, 2003), with some exceptions during the breeding season (Martínez-Abraín et al., 2008).

In the case of mammals, we could detect that Rodentia populations were slightly affected within few meters from infrastructure. By contrast, Artiodactyla species were affected up to distances of several hundred meters. Within Artiodactyla wild reindeer (*Rangifer tarandus*) was one of the most studied and sensitive species, with their population abundance being reduced up to several kilometres from infrastructure (Nellemann and Cameron, 1996; Nellemann et al., 2001, 2003a,b). These outcomes are consistent with the fact that small-sized mammals usually have smaller home ranges and migration distances compared to medium- and large-sized mammals, the latter being more sensitive to infrastructure development and habitat fragmentation (Harestad and Bunnell, 1979; Buskirk, 2009).

#### 4.3. Review limitations

Only some of the included studies used Before-After-Control-Impact (BACI) experimental designs, so we decided to use the largest reported distance in the study as control distance. Although the sensitivity analyses allowed us to remove some of the studies of lower quality and indeed the results did not vary in most of the cases, we acknowledge that our conclusions are restricted by the lack of proper comparators in some of the studies.

The identified publication bias is another of the weaknesses of our review. A few studies did not find negative effects of infrastructure on bird and mammal populations and were not included due to lack of proper data on the selected variables (e.g. Adams, 1984; Evans and Gates, 1997; Ballard et al., 2000); and there may exist many others that were never published due to non-significant results or that we were unable to obtain (grey literature). However, fail-safe numbers indicated that our results are sufficiently robust.

Many of the studies initially considered in our systematic review lacked suitable data for extraction and had to be left out of the analysis. Therefore we may have excluded potentially relevant studies and included lower quality studies due to availability of data on the selected variables.

The scope of the study was intended to be global and covered different types of biomes and habitats; nevertheless there is a geo-

graphical bias in the studies included in our review. Most studies were done in Europe and North America and therefore the applicability of the results to other geographic areas remains unknown.

#### 4.4. Conclusions and management implications

Changes in wildlife populations in the proximity of infrastructure have been reported for decades in a number of studies (see Table 2 in Supplementary material, available at <http://www.environmentalevidence.org/SR68.html>), and have been pointed out in relevant authors' reviews (Spellerberg, 1998; Trombulak and Frissell, 2000; Forman et al., 2003). Additionally, there have been previous attempts to quantify wildlife population decline in relation to distance from infrastructure, either locally (Forman and Deblinger, 2000) or at the global scale (UNEP, 2001; Nellemann et al., 2003b; Fahrig and Rytwinski, 2009), but none of them followed the guidelines for systematic reviews (Pullin and Stewart, 2006) or summarized the information by means of a meta-analysis. Our study represents a step forward within the field of road ecology research that may contribute to the understanding of the magnitude of the pernicious effects of infrastructure development on animal populations. Reported effects for most bird populations extend over distances up to about 1 km, and for most mammal populations up to about 5 km, with variation according to taxa and habitat type (Fig. 1 and 2). However, the evidence shown by our results is somewhat hampered by the limitations mentioned in Section 4.3. We therefore encourage researchers to perform BACI studies whenever possible and make their data available for researchers pursuing a systematic review. Should new studies that include these recommendations be released in the future, the review can be updated by including the new available evidence.

Although the patterns found in our analysis are clear, we would like to emphasise that these only represent a partial estimate of the actual effect of infrastructure on wildlife. Therefore, we highlight the importance of broadening the analysis to other taxonomic groups, such as herpetofauna, plants and invertebrates (e.g. Przybylski, 1979; Angold, 1997; Auerbach et al., 1997; Haskell, 2000; Shine et al., 2004; Barrows et al., 2006). Further research on these taxonomic groups would add up to the current models presented in this study, contributing to eventually produce a model that would provide an accurate estimate of the effects of infrastructure development on biodiversity.

The results of our meta-analysis will be implemented in the next version of the GLOBIO3 model, which is used to estimate the biodiversity loss at global, regional and national level at current state and for possible future scenarios and policy options (Alkemada et al., 2009). The results of the GLOBIO3 model have been reported in global assessments such as the second Global Biodiversity Outlook and the fourth Global environmental Outlook and are aimed to support policy makers on the elaboration of biodiversity conservation policies (sCBD and MNP, 2007; UNEP, 2007). The method is also used at the regional level (Verboom et al., 2007) and at the country level (e.g. in Viet Nam, Ecuador and Nicaragua).

Our study shows the importance of minimizing infrastructure development for wildlife conservation in relatively undisturbed areas. By combining actual species distributions with the effect distance functions we developed as a form of dose-effect relationship, regions sensitive to infrastructure development may be identified. More specifically, the effect distance functions can be used in models in support of decision making on infrastructure planning. This would mean a substantial improvement of the current situation in which, in most cases, results of previous studies on ecological impacts are barely taken into account (OECD, 2002; Roedenbeck et al., 2007).



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## Supplementary material

Supplementary material available at <http://www.environmental-evidence.org/SR68.html>. Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.biocon.2010.02.009.

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