RESEARCH ARTICLE

Ecological impact and cost-effectiveness of wildlife crossings in a highly fragmented landscape: a multi-method approach

Frans J. Sijtsma (ID) · **Eelke van der Veen** · **Arjen van Hinsberg** ·
Rogier Pouwels · **Renée Bekker** · **René E. van Dijk** · **Mark Grutters** ·
Raymond Klaassen · **Margriet Krijn** · **Maarten Mouissie** · **Eddy Wymenga**

Abstract

Context Road infrastructure construction is integral to economic development, but negatively affects biodiversity. To mitigate the negative impacts of

infrastructure, various types of wildlife crossings are realized worldwide, but little is known about their effectiveness, and cost-effectiveness.

Objective The paper contributes to the methodological and empirical discussion on the effectiveness of wildlife crossings for enhancing the quality of surrounding nature and its cost-effectiveness by analyzing a large-scale wildlife-crossings program in the Netherlands.

Method A multi-criteria cost–benefit analysis is applied, comprised of monetary and non-monetary measures, and a mixed-method approach is used to determine ecological effects. Ecological effects are expressed in the standardized weighted hectare measurement of threat-weighted ecological quality area (1 T-EQA = 1 ha of 100% ecological quality, averagely threatened). Cost-effectiveness is calculated comparing the monetary costs of intervention with ecological benefits (Euro costs/T-EQA), for different types of wildlife crossings and for two other nature policies.

Results The Dutch habitat defragmentation program has induced an increase in nature value of 1734 T-EQA at a cost of Euro 283 million. Ecological gains per hierarchically ordered groups of measures differ strongly: The most effective are ecoducts (wildlife crossing bridges) followed by shared-use viaducts and large fauna tunnels. Ecoducts generated the largest gain in nature value, but were also the most costly measures. In terms of cost-effectiveness, both large fauna tunnels and shared-use viaducts for traffic and animals outperformed ecoducts.

F. J. Sijtsma (✉) · E. van der Veen
Faculty of Spatial Sciences, University of Groningen,
Groningen, The Netherlands
e-mail: f.j.sijtsma@rug.nl

A. van Hinsberg
Netherlands Environmental Assessment Agency,
The Hague, The Netherlands

R. Pouwels
Wageningen Environmental Research, Wageningen, The Netherlands

R. Bekker
Dutch National Database Flora and Fauna (NDFF),
Nijmegen, The Netherlands

R. E. van Dijk · M. Grutters · M. Mouissie
Sweco, De Bilt, The Netherlands

R. Klaassen
Faculty of Science and Engineering, University of Groningen, Groningen, The Netherlands

M. Krijn · E. Wymenga
Altenburg en Wymenga, Feanwâlden, The Netherlands

Conclusions Ecoducts deliver ecologically, but their cost-effectiveness appears modest. Purchasing agricultural land for restoration of nature appears more cost-effective than building wildlife crossings. Yet, reducing environmental pressures or their effects on existing nature areas is likely to be most cost-effective.

Keywords Defragmentation of road infrastructure · Mixed-method evaluation · Ecoducts (wildlife crossing bridges) · Wildlife crossings · Cost-effectiveness · The Netherlands

Introduction

As economic development continues and human impacts on landscapes increase, biodiversity nearly always declines (Young et al. 2005; Li et al. 2010; Dirzo et al. 2014; Fraser et al. 2019). Infrastructure construction is part of economic development and may affect biodiversity and the ecological quality of landscape in different ways (Coffin 2007): direct habitat destruction, increased environmental pressure (e.g. exhaust or noise), and lower connectivity due to habitat fragmentation (van der Grift et al. 2009; Wilson et al. 2016; Torres et al. 2016; Sawaya et al. 2019). Decreasing habitat quality as well as reducing patch sizes leads to increases in local extinction of small populations and weaker connectivity leads to lower colonization of patches (Hanski 1998; Lindenmayer and Fischer 2006; Dennis et al. 2013). To mitigate the negative impact of infrastructure on habitats across Europe, Australia, Canada, and the U.S., different road defragmentation measures have been implemented to enable crossing of wildlife (Chruszcz et al. 2003; Bissonette and Cramer 2008; Mimet et al. 2016; Rytwinski et al. 2016).

However, Roedenbeck et al. (2007) argued that surprisingly little is known about the benefits and effectiveness of wildlife crossings from a conservation perspective. Taylor and Goldingay (2010) reviewed 244 studies on road and vehicle impact on wildlife. They found that, although wildlife crossings are becoming more common worldwide, their effects on populations are poorly described and there is an absence of cost–benefit analyses of road defragmentation measures. Moreover, while there is evidence that wildlife crossings may increase the use and

dispersal rates between patches to some extent (Sawaya et al. 2013; Soanes et al. 2015) an increase in actual use/dispersal is not direct evidence of higher population viability (Hodgson et al. 2011; van der Grift et al. 2013). Empirical studies focusing on population-level effects of wildlife crossings are scarce (van der Ree et al. 2009; van der Grift et al. 2013). Van der Grift and Pouwels (2006) and van der Ree et al. (2011) argued that assessing long-term viability of adjacent populations must be the most important parameter for measuring success. Several authors argued that it is imperative, given the large financial resources used to construct wildlife crossings, to conduct well-performed evaluations of the effects of these structures on long-term viability of adjacent wildlife populations (Corlatti et al. 2009; van der Ree et al. 2009; van der Grift et al. 2013). Earlier studies showed clearest effects when corridors connect the same habitat type (Eycott et al. 2010; Hodgson et al. 2011). Roedenbeck et al. (2007) and van der Grift et al. (2013) called for research designs with greater inferential strength and especially argue for either a manipulative or non-manipulative Before-After-Control-Impact (BACI) approach for most relevant road ecology questions as the preferred research designs.

Cost-effectiveness of road defragmentation investments is regularly called for, but the methodological prerequisites for enhancing cost-effectiveness are poorly addressed. If cost-effectiveness is a serious issue, then studies ideally would be able to allow comparisons of cost-effectiveness for different wildlife crossings in different regions and natural contexts, for different types of crossings, and for different nature policy instruments. Ovaskainen (2013) assumed on the basis of theoretical reasoning that construction of ecoducts (or wildlife crossing bridges) is more cost-effective than increasing the size of existing protected areas *only if* the ecoduct connects conservation areas larger than 500–5000 ha. But how can such a theoretical statement be empirically validated? Ecoducts typically cost millions of Euros, but a small fauna tunnel may require only much less, and a small fauna tunnel thus 'needs' far smaller ecological benefits to potentially outperform an ecoduct in terms of cost-effectiveness. So validation of Ovaskainen's statement requires comparisons of types of wildlife crossings as to their costs and benefits, while also comparing it to a completely different nature policy instrument: enlarging existing nature areas. To be able

to compare costs with benefits for this range of policy actions, it is crucial to apply a standardized measure for the biodiversity impacts of wildlife crossings. Such a measure should go beyond effectiveness for individual species which many studies limit themselves to (Fraser et al. 2019; Kormann et al. 2019), but instead should focus on the broad ecological impact in the surrounding areas of the crossings.

The aim of this paper is to gain insights into the effectiveness and cost-effectiveness of road defragmentation policies. Our case study is the Netherlands, which is interesting for two reasons. Firstly, because it has one of the densest infrastructure networks in Europe (Eurostat 2018). Secondly, the Netherlands has long been seen as a forerunner in the systematic planning and implementation of wildlife crossings to prevent and restore loss of habitat connectivity (van der Grift 2005), which, because there are many sites and structures to evaluate, makes for a good sample size. The defragmentation policy evaluated here is the Dutch Defragmentation Program 'Meerjarenprogramma Ontsnippering' (MJPO), which ran from 2005 to 2018 and sought to resolve 175[1] bottlenecks between existing national road and rail infrastructure (see Fig. 1) and the National Ecological Network (NEN).

In this paper we will contribute to filling the identified gaps in the literature by estimating the costs of realizing the wildlife crossings, by studying the ecological impacts with a mix of methods, and, by combining both, establishing the cost-effectiveness of wildlife crossings. To evaluate ecological impacts we adopt a so-called mixed-method approach using three different methods for assessing impacts at different spatial and temporal levels. Through the mix of methods for determining ecological impacts we will (1) assess model-wise the impact of the wildlife crossings on the habitat potential for supporting viable populations and we will (2) apply a non-manipulative BACI design that measures impact of the crossing on the surrounding nature via large scale species occurrence data, while, finally, we will (3) analyze available reports of wildlife crossings usage-monitoring to

establish actual use of the crossings. For the purpose of measuring effectiveness and cost-effectiveness analysis we will use the standardized weighted hectare T-EQA measurement: Threat Weighted Ecological Quality Area (Strijker et al. 2000; Sijtsma et al. 2011, 2013; van Puijenbroek et al. 2015). The choice for this broad measure of the ecological impact allows us to assess the effect(s) of wildlife crossings on the overall functioning of nature areas on both sides of the infrastructure link, rather than focus on mere use by a single species, or concentrate on reductions in wildlife-vehicle collisions. Moreover, using this metric allows us to explore a broader perspective on the effectiveness and cost-effectiveness of realising wildlife crossings by comparing tentatively and based on an earlier Dutch policy study (PBL & WUR 2017; van der Hoek et al. 2017), the MJPO measures to two other Dutch nature conservation measures: expanding nature areas or increasing the quality of existing nature areas by reducing environmental pressure.

Methods

MCCBA and a five-fold hierarchy of wildlife crossings

The general evaluation method (Fig. 2, Box 1) used in this study is the multi-criteria cost–benefit analysis (MCCBA) (Sijtsma 2006; Sijtsma et al. 2017). MCCBA takes the social cost–benefit analysis as its basis (Boardman et al. 2017), but enables the incorporation of not only monetary aspects (in our case the costs of constructing crossings), but also non-monetary criteria. This is especially important to our wildlife crossings evaluation, since ecological effects—the ultimate objective of the MJPO—are hard to capture monetarily. Once both types of calculations have been made, the cost-effectiveness is simply the ecological benefits divided by the costs. We examined the ecological benefits of five types of crossing structures using three different methods. Figure 2 gives an overview of the methodological steps used in this study. The different methodological boxes are numbered (1–10) to facilitate referencing in the text.

The MCCBA analysis is performed for the overall costs and benefits but benefits and costs are also aggregated throughout the analysis for different

---

[1] Rijkswaterstaat reports a total of 178 bottlenecks; however, for three of these no mitigation measures were known to be constructed in the database received in May 2017. For this reason, we examine 175 bottlenecks, 153 of which impact estimates were made using method 1 (see below).
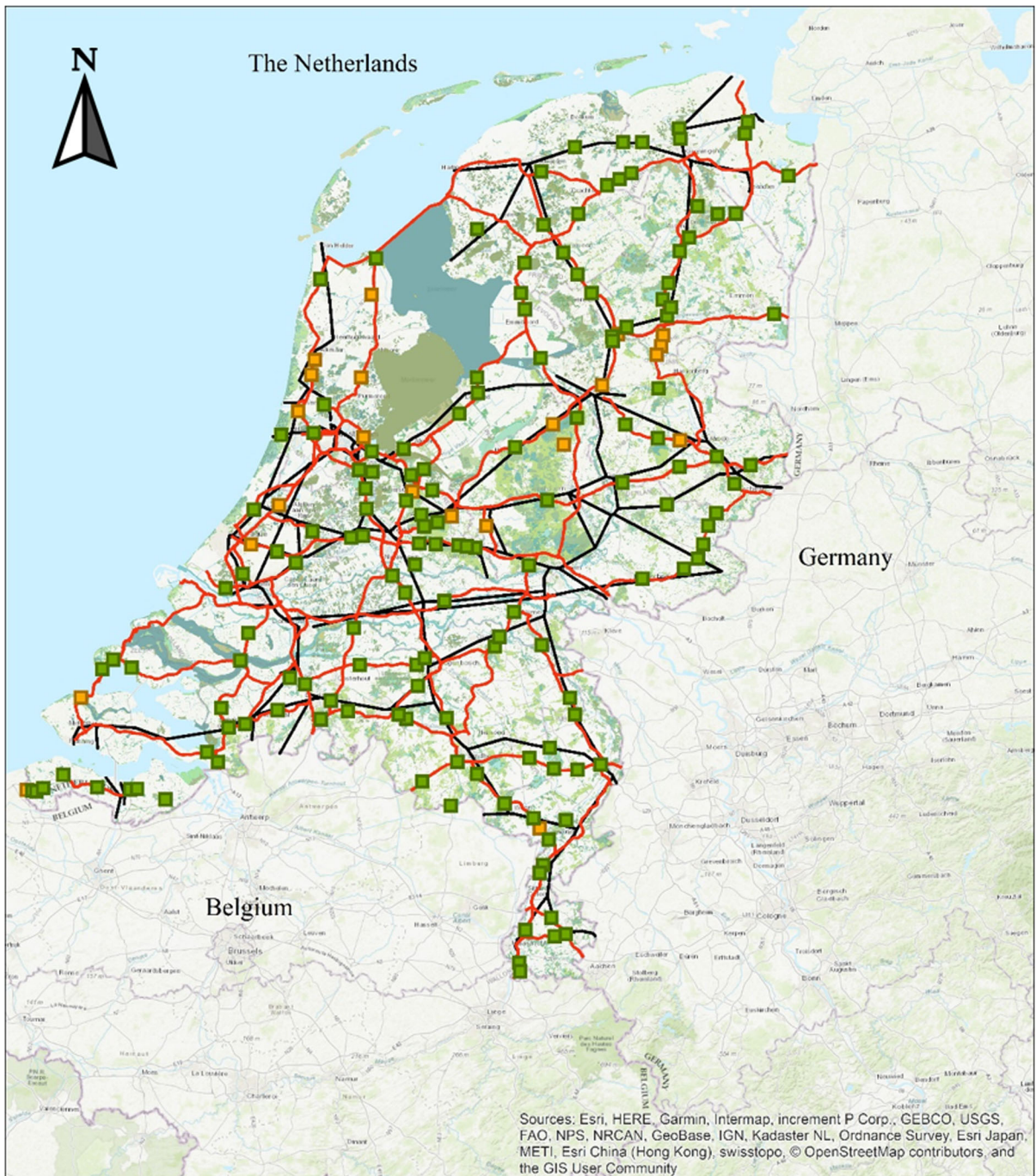
Fig. 1 Locations of the MJPO bottlenecks (n = 175). Green locations are used in our model-based method 1 approach (n = 153). Lines show the most important Dutch national highways (red) and railroads (black)

(combinations of) types of wildlife crossings, called hierarchy types (Fig. 2, Box 2). We use these hierarchy types because at one bottleneck location the wildlife crossings which are created to defragment infrastructure often combine different structures. For example, solving a bottleneck at one spot can mean that both a large and a few small wildlife tunnels are built. It is then hard to attribute the overall effect at one
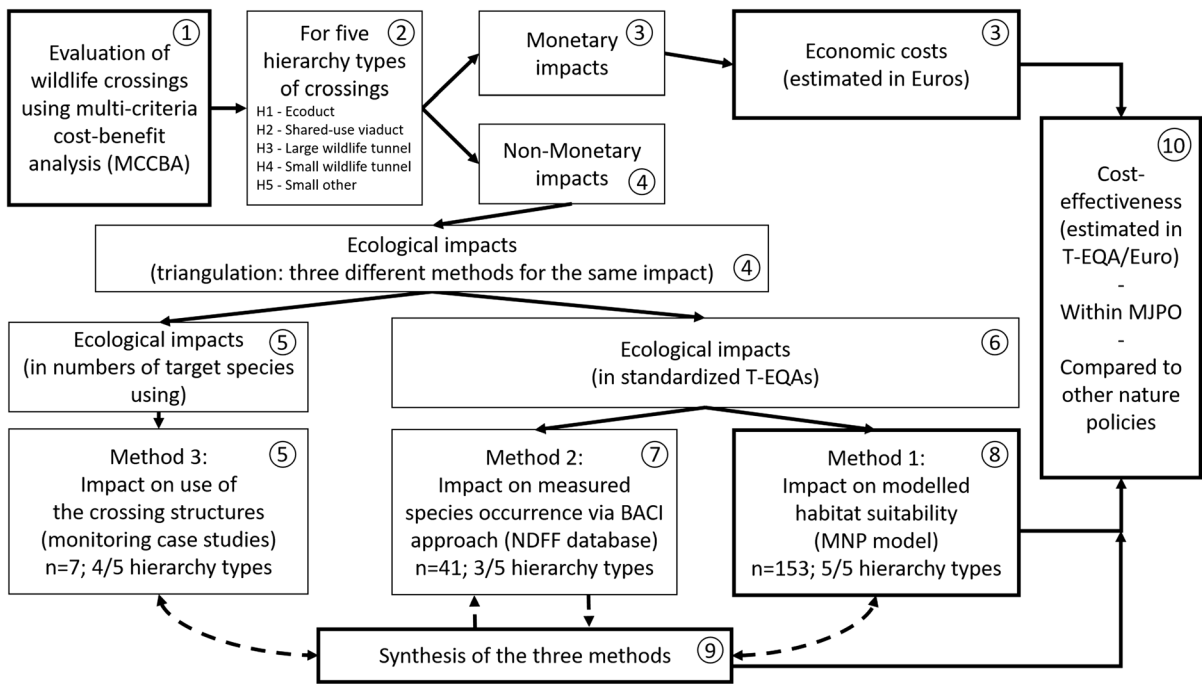
Fig. 2 The methodological structure of this study. Heavy-lined boxes (1, 3, 8, 9 and 10) show begin and end of the analysis, the light-lined boxes intermediate steps

location to a single measure. We therefore take all measures at one location together. Yet, the impact of measures on a location with only small tunnels will range less far, than the impact on a location with an ecoduct and some small tunnels. The centerpiece of our analysis is therefore a hierarchy of wildlife crossings (Table 1), through which we define the intervention(s) at one bottleneck by the highest level measure taken at that spot. The hierarchy is based on the potential number of species that can profit from the highest measure (Wansink et al. 2013). Ecoducts rank highest in the hierarchy, whereas small wildlife tunnels rank lowest. Descriptives of the five hierarchy clusters are shown in Table 1.

Costs of MJPO

The monetarily measured impacts in this study are the social costs of building the wild life crossings (Fig. 2, Box 3).These social costs of the program are estimated using the budget expenses provided by the Directorate-General for Public Works and Water Management (Rijkswaterstaat). The provided budget costs for an MJPO bottleneck may sometimes differ from its actual realization costs, but Rijkswaterstaat

experience shows that, on average, the budget costs provide valid estimates. Furthermore, wildlife structures may be entirely financed by Rijkswaterstaat, but other budgets may also provide funding (e.g. the national rail infrastructure organization or local municipalities). To acquire precise realization costs from the array of parties was beyond the scope of this research. We therefore *estimate* the investment needed for realizing the defragmentation measures via the available budget costs of Rijkswaterstaat, assuming that other organizations face similar costs. Based on availability of budget costs in the database provided by Rijkswaterstaat for 153 bottlenecks, we calculate average costs at a location for structures of the same type. The average costs were: Euro 5,791,000 for ecoducts (wildlife crossing bridges), Euro 794,000 for large fauna tunnels, Euro 574,000 for lattice fences and material, Euro 346,000 for culverts with continuous shores, Euro 231,000 for fauna step-out places, Euro 201,000 for shared-use viaducts, Euro 158,000 for small fauna tunnels, 92,000 for stub slopes with contiguous stubs and 52,000 for continuous shores under bridges. Since the database was incomplete and checking all entries was beyond the scope of our research, we used these averages as an estimation of

Table 1 A hierarchy of bottlenecks and wildlife crossings of the MJPO

Hierarchical rank	Name of hierarchical group	Number of species for which it potentially defragments	Number of bottleneck locations (based on hierarchical highest crossing structure)	Total number of crossing structures in MJPO program (often more than one structure at one bottleneck location)
1	H1-ecoduct	Highest	31	38
2	H2-shared-use viaduct	Lower than ecoducts	20	33
3	H3-large wildlife tunnel	Lower than viaduct	46	68
4	H4-small wildlife tunnel	Lower than large tunnels	65	264
5	H5-small other	Lowest	13	76
Total			175[a]	479[a]

[a]*Source* Rijkswaterstaat; numbers as at May, 2017

the costs *for each structure* of that type. We estimate the total costs *per bottleneck* by adding the average budget costs for all structure types realized at this bottleneck. Costs only include implementation costs of the program, not management and maintenance costs.

No discounting is applied. Recent guidelines for cost–benefit analysis in the Netherlands suggest a zero % discount rate, to which a risk premium can be added. Since the costs in this study lie in the past, we use a zero-risk premium, and the millions are actually nominal, non-inflation corrected, Euros.

Measuring ecological effects using three methods

Measuring biodiversity impacts monetarily is complex (Sijtsma et al. 2013), so we choose to measure non-monetarily and through a triangulation approach (Fig. 2, Boxes 4). Triangulation, or mixed method research, often combines quantitative and qualitative methods to study the same phenomenon (Jick 1979; Greene et al. 1989; Ivankova et al. 2006). To determine the ecological effects of the MJPO, we used three methods. Method 1 is model-based and assesses habitat potential for supporting viable populations. Method 2 is BACI-based and assesses ecological quality improvement on the basis of the largest and main Dutch database of species observations. Method 3 is case-study based and assesses the monitoring of species' usage of the structures as investigated in various MJPO (-related) monitoring reports.

Before describing these three methods in detail, it is important to comment on their combined logic, and identify their relative strengths and weaknesses. Mixed method research can broadly be seen as serving two distinct purposes (Jick 1979). The first is assuring richness of perspectives and explorations and the second is validating results. At the start of the research both purposes were considered equally valid, since the success of the three different methods was not clear at that time, but now that we report on the results the second, cross-validating purpose seems to dominate. This is also visualized in Fig. 2 by showing the dotted lines (below Boxes 5, 7 and 8) highlighting the synthesis of methods and the solid line from box 9 to 10 joining with the one from method 1 indicates that method 2 and 3 validated the ultimate use of the Model-based method 1. Still, both mixed-method purposes remain active and relevant throughout this research to some extent.

Method 1: Calculation on population viability based on model and expert estimates

Model-based method 1 (Fig. 2, Box 8) is used to determine the potential changes in viability of populations, and is essential for showing whether the spatial conditions allow sustainable survival of species with sufficient population size. Direct measurements of long-term population viability are not readily available, but models can help pinpoint species capable of forming viable populations, given the ecological conditions at a site. However, population models are

often site and species specific, which limits their use in an evaluation of national programs like MJPO. Model-based method 1 applies a national suitability model, the Model for Nature Policy (MNP), which is calibrated for use in the Netherlands (Pouwels et al. 2017). The MNP model links information on (1) environmental conditions, e.g., habitat type, nitrogen deposition, and ground water level with (2) the conditions required to ensure long-term viability of (meta)populations, e.g., size and amount of large habitat patches of 146 key bird, butterfly, and plant species of the Habitat and Bird Directives (Pouwels et al. 2017). The MNP model was used to calculate the number of species that might realise viable populations for each site in the Dutch National Ecological Network (Pouwels et al. 2017). The Methods appendix gives more background details on this model. Here we focus on the key operationalisations for MJPO. The MNP model in our study is applied to support the calculation of T-EQA-changes (Sijtsma et al. 2013) through wildlife crossings.

Formula (i) explains the steps needed for this T-EQA calculation (Fig. 2, Box 6).

$$\sum_{j=1}^{153} \Delta \text{TEQA}_{i,j}^{ha} = T_i * \Delta EQ_{i,j} * A_{i,j}^{ha} \qquad (i)$$

In Formula (i) the change in T-EQA^{ha} for a habitat i around bottleneck j is calculated by multiplying the change in ecological quality due to the applied measure (ΔEQ_{ij}) with the area of the habitat $\left(A_{i,j}^{ha} \right)$ and the threat-weight factor (T_i) of the habitat. Adding these changes for all 153 bottlenecks gives the total change. This calculation requires several steps.

Threat-weight The threat-weight factor (T) is a standard habitat-specific weight, based on the relative number of threatened species, calculated and published by the Netherlands Environmental Assessment Agency (Sijtsma et al. 2013) It expresses the vulnerability or threat value of an ecosystem at the spatial level of the Netherlands. For the Netherlands this factor ranges from 0.1 to 3.22. Extremely threatened ecosystems have the highest weight and commonly occurring ecosystem with common species have the lowest (Sijtsma et al. 2009; van Gaalen et al. 2014). The average weights of the list of ecosystems is 1 within the T-EQA

approach. Due to the threat-weight, wildlife crossings that positively affect threatened systems, deliver more T-EQA^{ha} than those affecting non-threatened systems.

Area Our second calculation is the Area factor (A^{ha}) of formula (i). To achieve an estimate of the ecological effects using T-EQA, one needs to determine which area is primarily influenced by a bottleneck. It is noteworthy that T-EQA and A in formula (i) are written with a superscript 'ha' for hectare. The 'Area' in a T-EQA calculation generally can be measured in any surface unit, but this choice will obviously affect the size of the T-EQA outcome. Here hectares are used, and the superscript 'ha' signals that.

Method 1 in this paper focuses exclusively on nature areas (N2000/National Nature Network) around bottlenecks, disregarding (i.e. subtracting from the buffer, see below) for instance built-up land and also agricultural land, which in the Netherlands, is often intensely farmed. By drawing a buffer around the location of a bottleneck, we capture the empirically most relevant direct impact area. Naturally, because of the differences in the empirical dispersion distances of different species, the main impact area of a wildlife crossing structure depends on the type of structure and the characteristics of its target species. We chose a generic approach to buffer sizes based on the hierarchy of Table 1. Importantly, a buffer should not be seen as the area to which the effects are limited, but rather as designated areas where the positive influence should be stronger than in (control) areas farther away. Choosing a buffer size involves balancing information on the distribution of dispersion distance of different species in the impacted ecosystems, but also the potential overlap between impact areas of different wildlife crossings. Also, since the Dutch landscape is very small scale, exceeding 2 km quite often indicates the occurrence of completely new land-use and influences.

Buffers were drawn in two sizes; one with a radius of 500 meters and the other with a radius of 2000 meters. A distance of 500 meters was thought suitable for small infrastructure crossings used by small and less mobile species. The empirical foundation on the dispersal distances of species that are sensitive to barrier effects is found in Verboom and Pouwels (2004) and van der Grift et al. (2009), who indicate that between 20 and 35% of the relevant Dutch species have a maximum dispersion distance of

500 m. A radius of 2000 m is used around large infrastructure crossings and is suitable to catch the impact on large, more mobile species. Using this calculation, a distance of 2000 m could be reached by 55–65% of the target species. We therefore choose a buffer distance of 500 m for small crossing structures (rank 4 and 5 in hierarchy), and 2000 m for large crossing structures (rank 1–3 in hierarchy).

As mentioned above, in the MJPO, multiple combinations of wildlife crossing structures are often constructed to solve a specific bottleneck, resulting in a 'cloud' of buffers that, when taken together, constitute the impact area of a bottleneck (Fig. 3). We use this cloud of buffers as the impact area of a bottleneck to determine the ecological effects.

Model-based method 1 includes 153 buffers of the 175 MJPO bottlenecks and thus excludes 22 bottlenecks. Of these, 13 had missing expert estimations for species that might profit from the wildlife crossings in van der Grift et al. (2009) (see below), and the other 9 were excluded due to lack of information on measures or because no nature was within the buffer areas.

Change in ecological quality Ecological quality (EQ) ranges from 0 to 100% and is commonly based on the percentage of species actually occurring, compared to an/the ideal reference situation. The Model for Nature Policy (MNP) model establishes the EQ on the basis of estimated sustainable occurrence (see below and the Methods appendix), and ideal occurrence of 146 key species per ecosystem type. If, for example, a forest has 25 potential species listed and MNP predicts that, with the given environmental and spatial conditions only 5 will sustainably occur, then that forest has an EQ of 20%. To reach a base estimate of EQ, an average of two scores is made, the first being the predicted vital species with both current environmental pressures and spatial conditions in place; the second is where all other environmental factors are assumed to be optimal, but restrictive spatial size conditions remain. This average may give a less realistic indicator for the short-term, but can also be seen as more realistic when successful environmental policies are anticipated for the future: policies that will lower environmental pressures.

The MNP model does not include the negative impacts of infrastructure, so estimating a change in ecological quality with the standard model output is not accurate enough. More detailed data by van der Grift et al. (2009) were therefore used to calibrate the model outcomes and include the effects of infrastructure. Out of a widely used list in Dutch Nature policy with 896 target species (Bal et al. 2001) van der Grift et al. (2009) established 88 species that were sensitive to infrastructure fragmentation. Those species are thus likely to benefit from wildlife crossings by MJPO. - Since the MNP model assesses environmental and spatial conditions for species without the explicit consideration of fragmentation by infrastructure, it was assumed that the EQ for those fragmentation sensitive species is 0 before the realization of the wildlife crossing and that the EQ will increase to the same value as the other species after: thus assessing the potential impact (given other environmental conditions). Per bottleneck and per surrounding ecosystems, percentages gain in species occurrence were estimated and applied to the EQ as estimated by MNP, delivering the $\Delta EQ_{i,j}$ in formula (i). For each
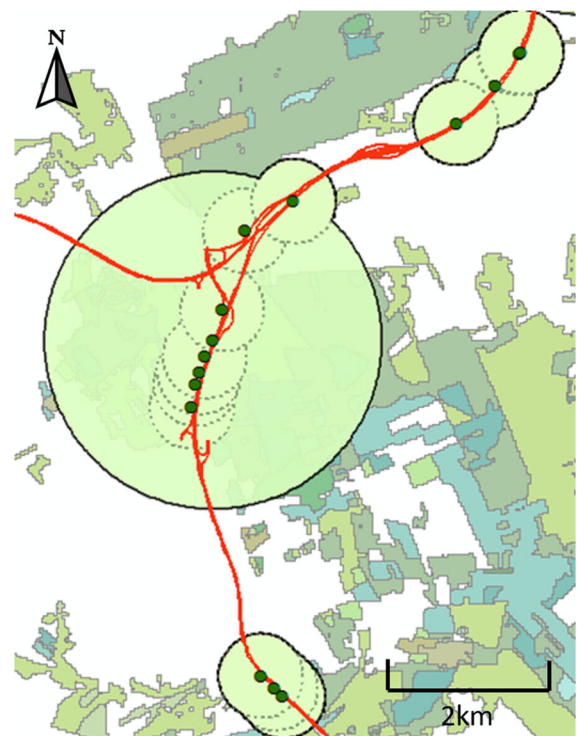

Fig. 3 'Clouds of buffers' as impact areas used in Model-based method 1. Red lines represent infrastructure. Different green and blue polygons reflect different habitat types. Green dots are locations of wildlife crossing structures. The black lines represent the cloud of buffers that dissolve into one impact cloud for a bottleneck; where dotted lines indicate the underlying individual structure buffers

ecosystem the fraction of species that is sensitive for infrastructure fragmentation and potentially occurring in that ecosystem is divided by all the target species that are potentially occurring in that ecosystem. In this way the percentage gain in EQ for a specific ecosystem takes into account that not all species in an ecosystem will profit from wildlife crossings as these species are not affected by infrastructure. The total EQ gain resulted from the sum of all affected ecosystem gains within the influence area of a bottleneck. Species using more than one ecosystem and profiting from wildlife crossings count as an improvement across all relevant ecosystems.

Method 2: BACI based on national database of species occurrence

BACI-based method 2 (Box 7) uses actual field observations (400.000 observations overall) of different species before and after the construction of wildlife crossings. This Method 2 also makes a T-EQA calculation, but relies on a different operational method. The Before-After-Control-Impact (BACI) design is claimed as the optimal study design for assessing human impacts on the environment (Roedenbeck et al. 2007; van der Grift et al. 2013). Method 2 uses a BACI design by compiling NDFF data (Dutch National Database on Flora and Fauna; www.ndff.nl/english/) from before and after road mitigation measures being taken. The NDFF, with 150 million validated species observations, is the largest comprehensive database on nature observations in the Netherlands. Until 2015 52% of the data came from protocolled observations (from monitoring schemes and structured (field) surveys), while 48% was opportunistic data, collected on an ad-hoc volunteer basis. Van Strien et al. (2013) have shown that also this second type of data can effectively be used for trend analysis (see Methods appendix for more details).

BACI-based method 2 calculations are made for a selection of 41 bottleneck sites and surrounding grids. The selection of bottlenecks had to be limited to the wildlife crossings of Hierarchy 1, 2 or 3 (main estimated impact area: 2 km radius) because NDFF data are mainly based on square kilometre grids. For the lower hierarchy measures, that have an impact area of 500 m radius, this would prevent differentiation between impact and control areas (see below). Bottleneck selection was further based on the final

year of construction being between 2006 and 2013. Since the NDFF proved to be best filled from 2001 onwards, the period 2006–2013 generated abundant NDFF observation data for 5 years before and 5 years after construction.

To select Impact and Control areas for each bottleneck location, three maps were confronted with each other: (1) the map with the core location of the wildlife crossing and a 2 km circle around it as the main impact area (2) the map of NDFF km^2 grid-cell map denoting the location of the species observations and (3) the map with nature types. On visual inspection four grids cells were chosen (preferably fully) within the impact radius and four grid cells outside the impact area. Thus for the BACI-based method 2, NDFF data were selected from four 1×1-km grids, e.g., two on either side of the road (or railway) and near the core of the wildlife crossing: the mitigation grid cells (see Fig. 4; squares within the light blue circle). As controls, another four 1×1 km grids, again two on either side of the road (or railway), are selected farther away from the core (see Fig. 4, the four fluorescent blue squares). They were chosen in a pairwise manner based on visual inspection and judgment of the representativeness and the similarity between impact and control areas in occurrence of nature types around each bottleneck. Control grids were chosen outside but as near as possible to the 2 km buffer. Figure 4 depicts an example of the selected grids around a bottleneck site.

The T-EQA calculations in Method 2 are made using T-EQA calculator software ('Natuurpuntencalculator 1.0') recently developed by the Dutch consultancy Sweco in close cooperation with scientists and stakeholders (Allema 2016; Jaspers et al. 2016) based on the same T-EQA documents and approach sketched above. The area calculation now differs somewhat because BACI-based method 2 works with nature areas in grid cells, not areas in dissolved buffers. Furthermore the Sweco T-EQA calculator also includes some agricultural areas alongside nature areas. The threat-weight, though for flexibility reasons calculated within the software, delivers the same Threat weight numbers as method 1 and is equal for an ecosystem type before and after (Allema 2016). The other two elements of T-EQA, area (A) and ecological quality (EQ) are calculated similarly for control and impact sites. With method 2, other than in method 1, the designation of the ecosystem types is determined

through the occurring species, thus allowing the area of ecosystems before and after to be different. Ecological quality is measured as the number of occurring species in an ecosystem compared to the ideal number of species (see Methods appendix for further details). The change in T-EQA points can therefore be a combination of area and ecological quality changes, and threat-weight if the ecosystem has changed before and after.

Sweco had before this study only applied the T-EQA calculation tool for relatively small NDFF data requests; also because data requests have to be paid for. However, specifically for this study, NDFF in cooperation with Rijkswaterstaat, granted permission for a nationwide data T-EQA request for a very large set of grids: for 328 (41×8) km^2 sites data were requested from the NDFF database, delivering in total, 393,000 species observations; on average 1200 per square kilometre grid (see Methods appendix for details).

To statistically test whether the difference in T-EQA before and after defragmenting the bottleneck differs between the mitigation and control areas, we use a Linear Mixed Model (using the package 'nlme' (Pinheiro et al. 2018) implemented in R (R Core Team 2018)). This model contains the difference in T-EQA as the dependent variable. The applied measure for solving the bottleneck is added as a fixed factor, consisting of the mitigating measure in three categories representing the hierarchy of mitigation bottlenecks: ecoduct, shared-use viaduct or large wildlife tunnel. A fixed factor indicating whether it concerns a mitigation or control area and a covariate for the number of observations used to calculate T-EQA were also included in the model.

Method 3: Qualitative case studies

The third method allows us to gain insights into the effectiveness of wildlife crossing structures and is based on literature and reports on monitoring actual use by target species. After an inventory of available monitoring studies, case studies were selected based on a set of criteria among which the representativeness
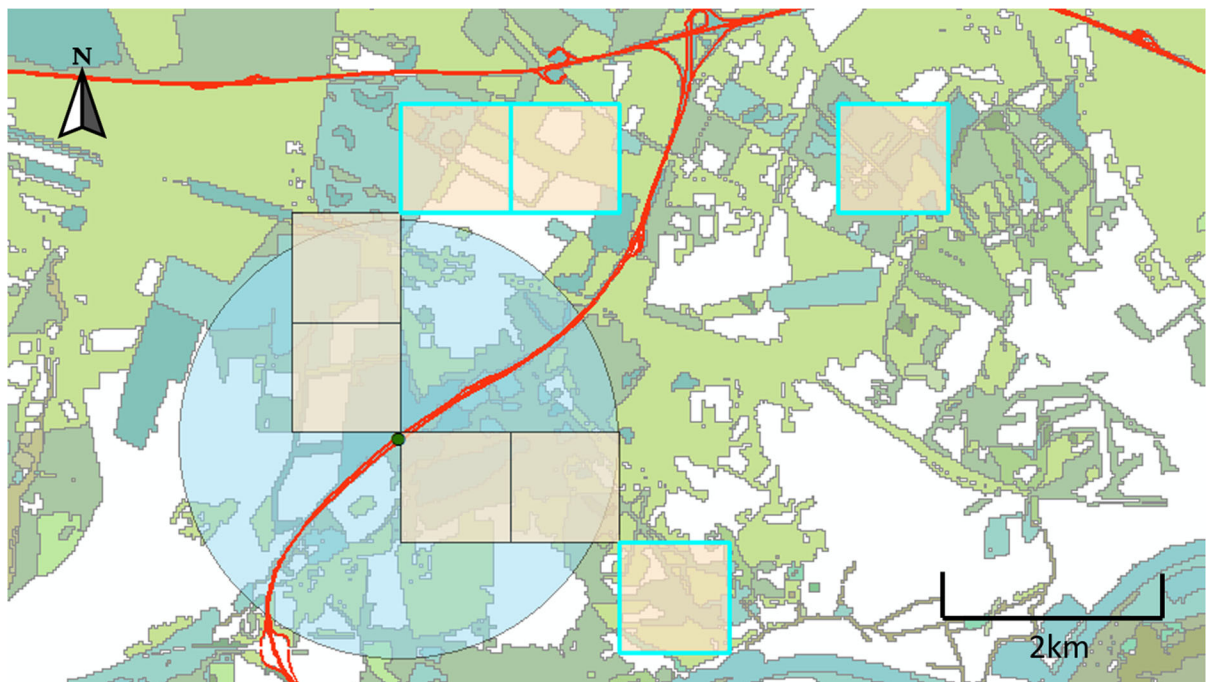

Fig. 4 Example of BACI-based method 2 grid selection. Red is infrastructure. The green point is the location of the wildlife crossing. The light blue circles shows the 2 km buffer area, where impact is expected more than farther away. In light yellow the eight 1x1 km grids are shown from which NDFF data are gathered: 4 mitigation/impact areas (with black lines; within the 2 km buffer) and 4 control areas (neon blue lines; outside the buffer). Impact and control grids are pair-wise on both sides of the road

for the programme, the spread across the hierarchy classes, and the availability of monitoring data over a preferred period of at least 5 years. Monitoring data proved to be available often for only 1 year (see Methods appendix for details). Monitoring efforts within the MJPO program have been limited in scope and time, allowing us to initially select 13 cases (see Methods appendix for background details). Upon perusal, we identified factors which might explain the measured effectiveness, such as local characteristics of the measure or its neighbourhood (Krijn and Wymenga 2018). For the purpose of validating the other methods we then focused on the available quantitative information in these case studies. For 7 cases effectiveness could be calculated by us by quantifying the actual use by a number of target species as a proportion of the specified target species that could potentially use the structure.

Synthesis and two types of cost-effectiveness analysis

Synthesis

Three methods were used to establish the ecological impacts of wildlife crossings. The next step was to synthesize the outcomes (Fig. 2, Box 9). All three methods have their own merits and insights to offer, but the quantitative scores per hierarchy group seems to offer the clearest 'linking pin' across all methods. The results of the Case-based method 3 are only moderately comparable to the other two methods. Methods 1 and 2 use the same T-EQA concept, which is strong for comparing, but they have a different number of bottlenecks, and they also differ in the number of hectares involved. Method 1 uses extended clouds of 2 km buffers, while method 2 uses only the 4 km^2 grids cells within the 2 km buffers and method 2 regularly has different areas of ecosystems before and after realisation of the wildlife crossings. Furthermore, method 2 also involves some agricultural lands, whereas method 1 is restricted to nature areas. These difference in operationalisation of the T-EQA measurement prevents one-on-one comparison of the absolute T-EQA values.

However, the relative patterns in the height of the T-EQA scores per hierarchy class of wildlife crossings (i.e. 4, 10 and 8 is a matching pattern with 8, 20 and 16, while 10, 2 and 7 are not) does give a clear validation

option and will be applied. This step will also provide a validation for the best method to perform cost-effectiveness analysis (Box 10): if the results of method 1, 2 (and to a lesser extent method 3; since it is a non-T-EQA score) validate each other, then the Model-based method 1 will be the preferred method since it has the widest reach (153 vs 41 or 7 bottlenecks and 5 versus 3 or 4 hierarchy groups). If there is wide disparity then the BACI design and observation-based ground truthing of method 2 seems to be empirically strongest. In this case the complementary richness of different methods also comes more to the fore.

Cost-effectiveness analysis

The first part of the cost-effectiveness analysis (Box 10) is about the of hierarchy groups within MJPO. In this part of the analysis the ecological gain (absolute change in T-EQA) totalled for the different hierarchy groups (ecoducts/large wildlife tunnels/and shared-use viaduct etc.) are confronted with the monetary cost: thus T-EQAha per million Euros.

The second part of the cost-effectiveness is about comparing MJPO to other nature policies. Few studies have compared the (cost-)effectiveness of fragmentation measures in relation to other types of conservation strategies. Lawton et al. (2010) conducted an extensive analysis of the range of measures for restoring biodiversity in England. They suggest a hierarchy of benefits in which the order is from high to low: (1) better management of existing sites, (2) realization of bigger sites, (3) realizing more sites, (4) enhancement of connectivity and (5) the creation of new corridors. Ovaskainen (2012) agrees, but says cost-effectiveness is dependent on the degree of connectivity achievable and the size and habitat types of the areas connected. We extend this discussion by exploring the relative effectiveness and cost-effectiveness of road infrastructure defragmentation compared to other nature conservation measures. For this comparison, we make use of policy report by PBL & WUR (2017) which uses the exact same MNP model as our method 1 and also assesses cost items in a comparable way. The report analyzes the costs and benefits of the Dutch 'Nature Pact' policy, which aims to conserve biodiversity through two main policies: expanding and managing the National Ecological Network, and increasing the quality of its existing areas. We calculated T-EQA for these two main policies based

Table 2 MCCBA evaluation: overview of cost, benefits and cost-effectiveness of the MJPO wildlife crossings program

MCCBA—MJPO		Total	Hierarchy group				
			H1-ecoduct	H2-shared-use viaduct	H3-large wildlife tunnel	H4-small wildlife tunnel	H5-small other
Part 1	Total costs (million Euro)	283	194	16	33	36	3
	Percentage	100%	69%	6%	12%	13%	1.2%
	Number of bottlenecks	(n = 153)	(n = 26)	(n = 20)	(n = 44)	(n = 56)	(n = 7)
Part 2	Ecological effects						
	Method 1						
	T-EQA change—model based viability	1734	1031	186	422	92	3
	Percentage	100%	59%	11%	24%	5%	0.2%
	Number of bottlenecks	(n = 153)	(n = 26)	(n = 20)	(n = 44)	(n = 56)	(n = 7)
	Method 2						
	T-EQA change—BACI based observations	608	380	53	175		
	Percentage	100%	63%	9%	29%		
	Number of bottlenecks	(n = 41)	(n = 13)	(n = 6)	(n = 22)		
	Method 3						
	Actual use of corridors (% of species found)		62%	20%	7%	0.0%	
	Number of bottlenecks	(n = 7)	(n = 2)	(n = 2)	(n = 2)	(n = 1)	
Part 3	Cost-effectiveness						
	T-EQA/million Euro (based on Method 1)	6.1	5.3	11.4	12.6	2.6	0.8

on the analyses by PBL & WUR (2017; data kindly provided by the PBL & WUR report authors; see the Methods appendix for the details).

Results

Overview

Table 2 provides an overview of our MCCBA analysis results by combining both monetary costs and non-monetary nature benefits expressed in T-EQA^{ha}. Part 1 of the table provides the estimate of the economic costs of the MJPO (for comparison reasons based on the 153 bottlenecks of method 1); part 2 summarizes results of the three methods; and part 3 shows the cost-effectiveness analysis. The total costs of MJPO were estimated at €Euros 283 million. Table 2 indicates that the highest costs by far were in the ecoducts hierarchical group, at 69% (€Euros 194 million) of the program's total cost, followed by 13% and 12%, respectively, for small wildlife tunnels and large wildlife tunnels. Shared-use viaducts accounted for 6% of the costs, and the remaining structures comprised 1.2% of the total costs.

The main ecological results are observable via method 1. The total gain in T-EQA^{ha} is 1734. The 26 ecoduct bottlenecks contribute to the largest gain in T-EQA^{ha}: 1031, or 59%. The table shows a 5% T-EQA gain for the 56 small wildlife tunnels, whereas large wildlife tunnels and shared-use viaducts contribute 24% and 11%, respectively.

Method 2 used 41 bottlenecks (of hierarchy 1–3) and resulted in a T-EQA change of 608. A large part of this change (63%) can be contributed to ecoducts. Statistical analysis of results of this BACI-based method 2 using a Linear Mixed Model found that the difference in T-EQA between before and after resolving a bottleneck is significantly smaller for control sites than for sites with mitigation measures. This difference was on average 3.7 (± 1.3 SE) T-EQA smaller for control sites than for mitigated sites (Table 3). Table 3 also shows that T-EQA is significantly different among the various hierarchy groups, with higher T-EQAs observable at higher hierarchy types (see also Methods appendix).

Returning to Table 2, there we see the quantitative results of Case-based method 3 on the use of corridors by target species. Within the MJPO, 13% of the 175

bottlenecks and 6% of 479 wildlife crossings were actually monitored in the field regarding their use. The case study result percentages in Table 2 only show the proportion of pre-specified target species that used the corridors corresponding to the different hierarchy groups. Our calculations here are based on seven case studies. Ecoducts appear to function relatively well in terms of actual movements, with 62% of the targeted species actually using the structure. For instance, the ecoduct Tolhuis at the Veluwe was reported to be used quickly by wild boar and red deer, and monitoring showed 45 birds using the structure, 24 of which were breeding birds. It is noteworthy that the primary focus of our case studies was to obtain additional *qualitative* insights and not to quantify their effectiveness.

Qualitatively, the case studies show that none of the monitoring studies within the MJPO focus on reduced mortality, removing barrier effects, or viability of populations. Instead, monitoring focuses on actual use of wildlife crossings by target species. Various monitoring documents mention the dimensions of a wildlife crossing and its design as important factors of effectiveness. For example, the guiding measures (like fences) on large wildlife crossings are mentioned as a possible reason why fewer mobile species (e.g. amphibians) were spotted at some crossings. Maintenance and management of crossing structures and their proximity to other defragmenting structures is also often identified as a determinant for successful use.

Synthesis

Now that results of all three methods are clear, how do they relate? Figure 5 shows that, per bottleneck, these operationally completely different methods show quite similar results regarding the relative scores per bottleneck: the pattern is quite similar. Only ecoducts score relatively high with method 1. Through this result, BACI-based method 2 seems to largely validate Model-based method 1, which may be realistic since Method 1 also estimates long-term potential. Because of the limited quantitative power of the Case-based method 3 (n = 7), it is not shown in Fig. 6. But the results from method 3 in Table 2 appear to validate the relative pattern of outcomes of the ecoducts and shared-use viaducts.

Because of its broadest range (n = 153) Model-based method 1 is most suitable for overall MJPO evaluation purposes. Based on the pattern of scores per

Table 3 Results of the Linear Mixed Model with the difference in T-EQA before and after resolving the bottleneck as the dependent variable

Variable	Coefficient $\beta \pm$ SE	t	P
Difference T-EQA before after for control minus impact areas	$- 3.73 \pm 1.31$	$- 2.9$	0.007
Average T-EQA change of lowering hierarchy type	$- 3.66 \pm 1.43$	$- 2.6$	0.014
Number of observations used to calculate T-EQA	0.000611 ± 0.000	3.4	0.002

N = 82 (41 mitigation/impact areas and 41 control areas)
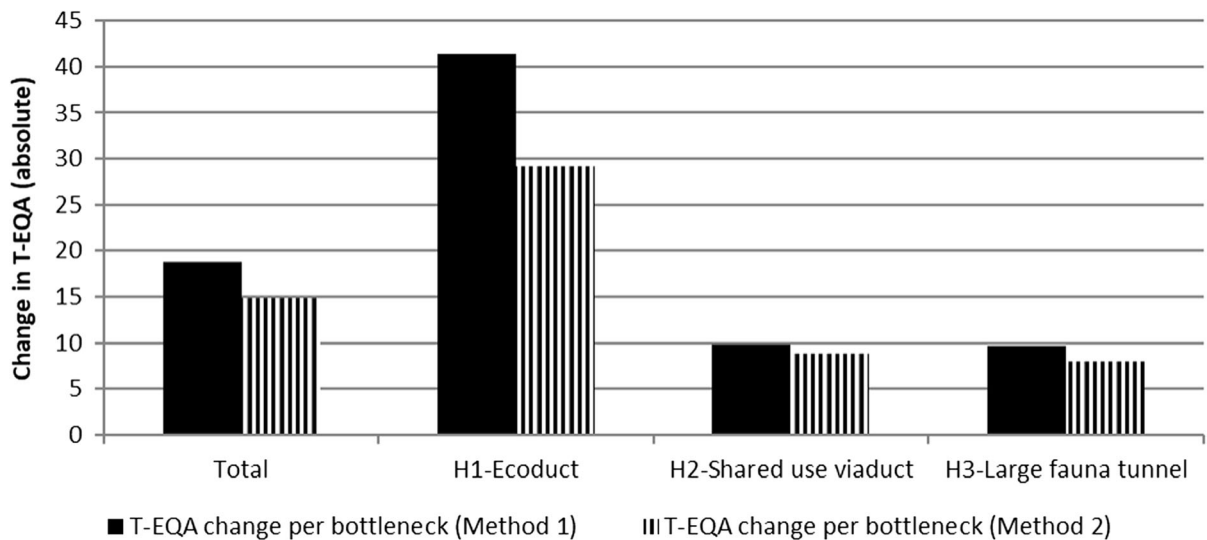
Fig. 5 Comparison of T-EQA change results of Model-based method 1 and BACI-based method 2

Fig. 6 Cost-effectiveness of three different nature policy strategies in T-EQA per million Euros
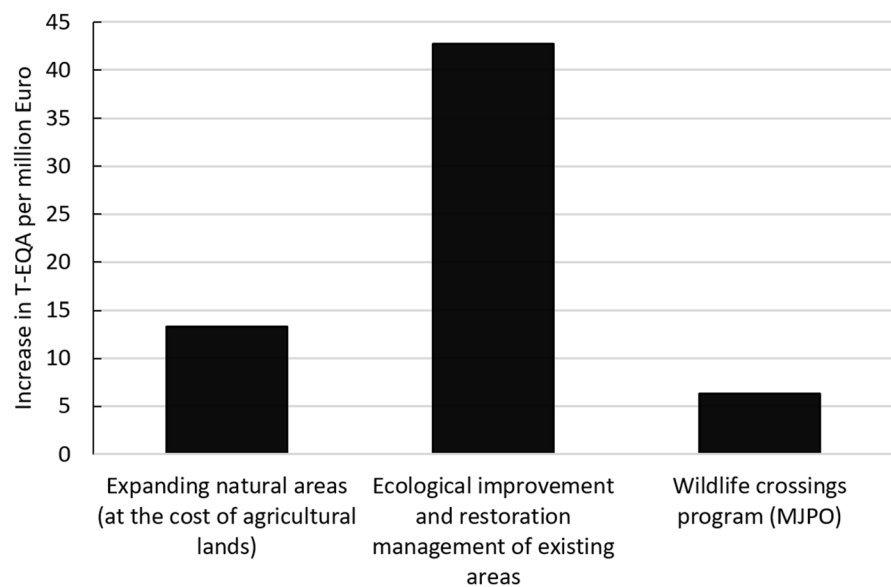
hierarchy we find enough support for using Model-based method 1 results as the main results.

Cost-effectiveness within MJPO

Returning to (the last part of) Table 2, we can now look at part 3 of the MCCBA: the cost-effectiveness. This part shows the relative T-EQA^{ha} per million Euros, and it is based on the method 1 results. Overall, an investment of € 1 million in an MJPO wildlife crossing yields 6.1 T-EQA. The results make clear that while ecoducts are the biggest contributors to biodiversity increases in absolute terms, with a cost-effectiveness of 5.3 T-EQA/€mln, ecoducts are considerably less cost-effective than the large wildlife tunnel (12.6 T-EQA/€mln) and shared-use viaduct (11.4 T-EQA/€mln) groups.

Effectiveness and cost-effectiveness of MJPO compared to other nature policies

Figure 6 show the results for the cost-effectiveness analysis of MJPO compared to two other nature policies, which are part of the plan of the so-called Nature pact. To recall, this analysis uses a policy report and underlying data by PBL & WUR (2017) which uses the exact same MNP model as our method 1 and also assesses cost items in a comparable way. Expanding natural areas according to the Nature pact plan would result in an increase in T-EQA due to the expansion of 41.001 T-EQA. Ecological improvement and restoration of existing areas would induce a gain in T-EQA of 58.413. These are large numbers of absolute gains compared to the 1734 of the MJPO (compare Table 2).

Realization costs were calculated in a similar fashion as above for MJPO: no maintenance and management costs were assessed (derived from PBL-WUR 2017; see Methods section). We calculated costs of Euros 3.08 billion for planned expansion of natural areas between 2010 and 2030, and for improving existing ecological areas in that period: Euros 1.37 billion. These too are very large absolute numbers compared to the Euro 283 million spent on MJPO. Using these numbers, Fig. 6 shows the cost-effectiveness of the Nature pact strategies compared to the MJPO.

Discussion

Effectiveness and cost-effectiveness through mixed-method research

In this study we applied different calculations of the effectiveness and cost-effectiveness of wildlife crossings. Nature quality in general terms has increased due to wildlife crossings around bottlenecks as habitats have become more suitable for viable population sizes of targeted species. All methods in this research show the ecological effectiveness of ecoducts (wildlife crossing bridges): relatively large ecological impacts for the ecoduct hierarchy group was seen in all three methods. Ecoducts were frequently used by a relatively broad range of species, according to monitoring reports (Case-based method 3). Analysis of 400,000 real-life observations show that barrier effects were reduced (BACI-based method 2), and ecoducts positively influenced the long-term habitat potential for supporting viable populations (Model-based method 1).

Lesbarreres and Fahrig (2012, p. 375) assert that monitoring research on the effectiveness of ecopassages often yields 'equivocal or weak results' which are unique to a specific location and cannot be compared to a baseline or benchmark. Case-based method 3 confirms this practice. With respect to the direct monitoring within the MJPO program, method 3 made clear that available data were limited, as only 13% of the individual bottlenecks had been monitored, and none of the monitoring studies investigated changes in population viability. Moreover, in almost all cases there was no baseline situation for monitoring. The use of crossing structures was generally monitored over a short period (i.e., 6 weeks) in spring and/or autumn, to coincide with most migration movements (van der Grift 2010; van der Grift and van der Ree 2015). Monitoring methods varied with regard to target species, thereby hampering comparability. Monitoring results are hard to extrapolate to the entire program and contain limited advice for other programs.

In this context, Glista et al. (2009, p. 5) suggest that 'the efficiency of road mortality mitigation approaches should be determined via a post-implementation monitoring', while Roedenbeck et al. (2007) observe the rarity of long-term monitoring of measuring changes in populations. Our study indicates that

post-implementation monitoring may have a different design than that expected by the abovementioned authors. Assessing generally available species observations from various sources can—if available like in the Netherlands—also work for post-implementation monitoring (method 2). Roedenbeck et al. (2007) call for research designs with greater inferential strength that allow for generalizable and robust conclusions; they attest to either a manipulative or non-manipulative BACI approach for most relevant road ecology questions as the preferred research designs. Our study used a non-manipulative BACI design in method 2, and in that sense delivers on the demand of Roedenbeck et al. However, we also adopted a mixed-method approach to maximize inferential strength. Three separate ecological teams were organized (Method 1, author researchers Pouwels/van Hinsberg; Method 2, author researchers van Dijk/Grutters/Mouissie/Bekker; Method 3, author researchers Krijn/Wymenga), which separately but concertedly researched ecological effectiveness from three different angles and approaches. A common component, however, was the use of the common metric T-EQA, for two of the three methods (Sijtsma et al. 2011, 2013; van Puijenbroek et al. 2015), allowing limited though serious validation of results. Without such a metric, no comparative effectiveness within this study, or cost-effectiveness compared to other nature policies (see below) could have been reached. Using the T-EQA metric puts the focus on the change in the quality of surrounding nature, measured by the change in occurrence of all species relevant and significant to the affected ecosystems. It thus requires a shift away from only assessing the effectiveness for single species which has been investigated in several studies (e.g. Chruszcz et al. 2003; Soanes et al. 2015; Sawaya et al. 2019). We feel such an approach, if adopted more widely in other studies, could strongly enhance 'inferential strength that allow[s] for generalizable and robust conclusions' called for by Roedenbeck et al. (2007).

Comparing results on effectiveness and cost-effectiveness

Strategies to mitigate the impacts of roads on wildlife have been studied by Jackson and Griffin (2000) and an overview of the effectiveness of corridors has been presented by Gilbert-Norton et al. (2010). Cost-effectiveness has been researched much less. A telling

example is the study by Sawaya et al. (2013). In their Banff National Park study they do show effectiveness of crossings for bears at the population level. But although they explicitly observe that much of the "debate has understandably focused on whether wildlife crossings (…) are worth the relatively high costs" (Sawaya et al. 2013, p. 722), they only implicitly suggest that their effectiveness results help the mentioned cost-effectiveness discussion. In our study we explicitly address cost-effectiveness. A key result of our study is that the cost-effectiveness of ecoducts is substantially less than large wildlife tunnels and shared-use viaducts (less than half) (5.3 vs 12.6 and 11.4). This is not because ecoducts do not deliver ecologically, but mainly because they are far more costly than large wildlife tunnels. Perhaps with the same amount of money more than twice as much nature could be realized through the use of shared-use viaducts and large wildlife tunnels. However, comparisons of cost-effectiveness using T-EQA presuppose the exchangeability of different nature types and species, which is not always a realistic possibility.

Variability within hierarchical groups is also an issue. An important factor determining the variation in effectiveness is the position of a wildlife crossing measure in the landscape. All three methods suggest that (cost-)effectiveness increases when crossings are constructed close to large (and/or high quality) nature areas. Proximity to natural areas in method 3 case studies was mentioned as an important explanation for data on species crossings. From the BACI measurements a positive correlation between T-EQA before and after the mitigation measures was found, with the largest effect in areas with the highest nature quality. This is shown in Fig. 7. The steeper slope of the black line for sites next to the mitigation measures, compared to the slope of the green line for the control areas, indicates the positive effect, while the increasing distance between the lines at larger T-EQAs signifies a larger effect at areas with higher nature quality.

Comparing wildlife crossings with other conservation strategies

Given the large scale impact of infrastructure on the surrounding nature even at relatively great distances (Torres et al. 2016), it is important to understand the value of wildlife crossings within the range of possible
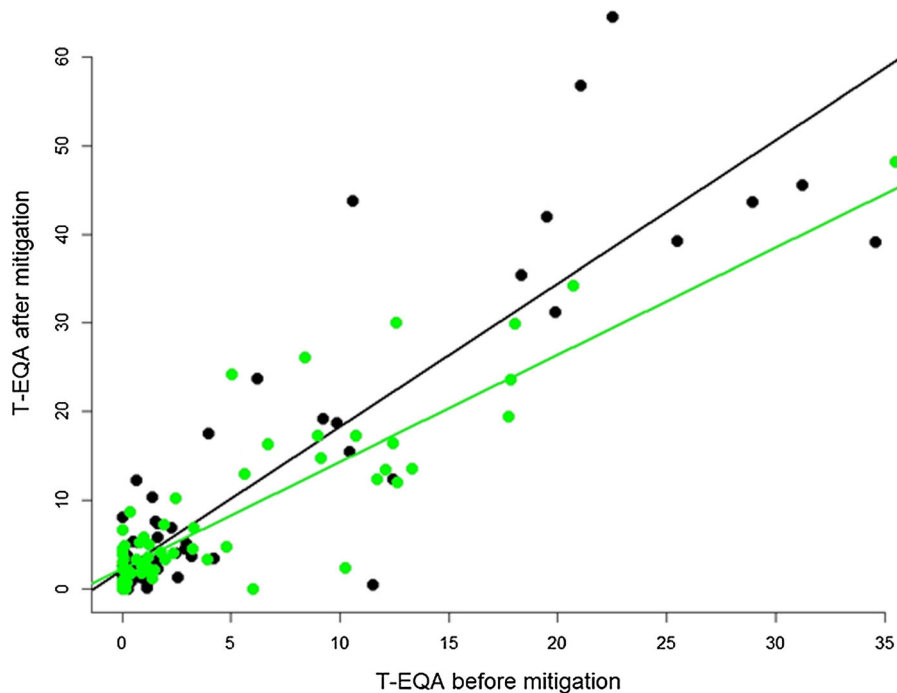
Fig. 7 The correlation between the T-EQAs before and after mitigation for both mitigation areas (in black) and control areas (no mitigation, in green)

nature policy options. The results presented in Fig. 6 on the cost-effectiveness of MJPO compared to Nature pact policies that aim at either expanding natural areas (at the cost of agricultural lands) or improving the ecological quality of existing nature areas through restoration management clearly show that improving the quality of existing areas by ecosystem restoration and environmental improvement (e.g., raising ground-water level) is the most cost-efficient strategy. According to Lawton et al. (2010), restoring connectivity between nature areas is expected to be the least cost-effective compared to improving the quality of nature areas, enlarging nature areas, and realizing more nature areas. Ovaskainen (2012) agrees with Lawton et al. (2010), although he argues that cost-effectiveness is dependent on the degree of connectivity achievable and the size and habitat types of the connected areas. Our analysis provides some empirical support for both Lawton and Ovaskainen.

However, these findings may require refinement, as our results show that effectiveness of wildlife crossing measures also depends on the surrounding landscape. The variance in cost-effectiveness of MJPO measures is also rather high; a large part of the MJPO measures has a cost-effectiveness lower than 10 T-EQA per

million, while a small set scores between 20 and 40 T-EQA per million. As such, some part of MJPO measures can be more effective than expanding natural areas and/or taking restoration measures; local context including the amount of large scale nature nearby, is critical. Issues of availability of space and time for actions other than wildlife crossings should also be mentioned. Wildlife crossing measures are often relatively simple and quickly spatially feasible around national infrastructure because government owns most of the land, whereas land elsewhere must be bought and this can be time-consuming. Moreover, measures directed at infrastructure can often be implemented simultaneously with the building of the infrastructure (e.g., road construction or widening), but (re)development of various natural habitats on former agricultural lands can take decades (Wiertz et al. 2007).

Conclusions

This study evaluated the Dutch road infrastructure defragmentation Program 'Meerjarenprogramma Ontsnippering' (MJPO), which ran from 2005 to

2018. The program delivered 1734 T-EQA at a cost of Euro 283 million. To clarify: 1734 T-EQA means the program created the equivalent of 1734 new nature hectares with 100% ecological quality (of ecosystems with an average threat weight), or improving the equivalent of 3468 hectares with 50% ecological quality. From a scientific point of view it is hard to make claims about whether such a gain is worth the Euro 283 million. We did see that this cost-effectiveness seems to be far lower than nature policies aimed at expanding natural areas or improving the ecological quality of existing nature areas through restoration management. However, what is also clear is that compared to the huge debates and conflicts which generally surround these other Dutch nature policy instruments, the MJPO program delivered with relative ease and largely within the set time frame. Ecological delivery and little public resistance seems to warrant consideration of a future continuation of wildlife crossing policy, but perhaps more attention could be given to the most cost-effective elements of the program: the large wildlife tunnels and the shared-use viaducts and less to the ecoducts (wildlife crossing bridges).

Furthermore, this study aimed to contribute to the methodological and empirical discussion on the effectiveness and cost-effectiveness of wildlife crossings. It showed that monitoring and the analysis of the effectiveness of wildlife crossings need not depend on case-specific usage monitoring, but that both the use of nationwide large scale species occurrence data as well as a habitat suitability model can assess the impact. Since, in the age of big data and easy digital sharing of models, these approaches can probably be relatively easily reproduced for other locations, our study opens the way to more international comparative research on (cost-)effectiveness of wildlife crossings.

References

Allema (2016) Natuurpuntencalculator 1.0 ('Nature point (= T-EQA) calculator 1.0'). December 2016. Sweco, Houten

Bal D, Beije HM, Fellinger M, Haveman R, Van Opstal AJFM, Van Zadelhoff FJ (2001) Handboek natuurdoeltypen. Expertisecentrum LNV, Wageningen

Bissonette JA, Cramer PC (2008) Evaluation of the use and effectiveness of wildlife crossings. National Cooperative Highway Research Program (NCHRP) report 615. Transportation Research Board, Washington

Boardman AE, Greenberg DH, Vining AR, Weimer DL (2017) Cost-benefit analysis: concepts and practice. Cambridge University Press, Cambridge

Chruszcz B, Clevenger AP, Gunson KE, Gibeau ML (2003) Relationships among grizzly bears, highways, and habitat in the Banff-Bow Valley, Alberta, Canada. Can J Zool 81(8):1378–1391

Coffin AW (2007) From roadkill to road ecology: a review of the ecological effects of roads. J Transp Geogr 15(5):396–406

Corlatti L, Hacklaender K, Frey-Roos F (2009) Ability of wildlife overpasses to provide connectivity and prevent genetic isolation. Conserv Biol 23(3):548–556

Dennis RL, Dapporto L, Dover JW, Shreeve TG (2013) Corridors and barriers in biodiversity conservation: a novel resource-based habitat perspective for butterflies. Biodivers Conserv 22(12):2709–2734

Dirzo R, Young HS, Galetti M, Ceballos G, Isaac NJ, Collen B (2014) Defaunation in the Anthropocene. Science 345(6195):401–406

Eurostat (2018) Inland transport infrastructure at regional level. Eurostat—Statistics Explained. Data retrieved April 2018. https://ec.europa.eu/eurostat/statistics-explained/index.php/Inland_transport_infrastructure_at_regional_level#Overview

Eycott AE, Watts K, Brandt G, Buyung-Ali LM, Bowler D, Stewart GB, Pullin AS (2010) Do landscape matrix features affect species movement. CEE Review: 08–006

Fraser DL, Ironside K, Wayne RK, Boydston EE (2019) Connectivity of mule deer (Odocoileus hemionus) populations in a highly fragmented urban landscape. Landsc Ecol 34(5):1097–1115. https://doi.org/10.1007/s10980-019-00824-9

Gilbert-Norton L, Wilson R, Stevens JR, Beard KH (2010) A meta-analytic review of corridor effectiveness. Conserv Biol 24(3):660–668

Glista DJ, DeVault TL, DeWoody JA (2009) A review of mitigation measures for reducing wildlife mortality on roadways. Landsc Urban Plan 91(1):1–7

Greene JC, Caracelli VJ, Graham WF (1989) Toward a conceptual framework for mixed-method evaluation designs. Educ Eval Policy Anal 11(3):255–274

Hanski I (1998) Metapopulation dynamics. Nature 396(6706):41–49

Hodgson JA, Moilanen A, Wintle BA, Thomas CD (2011) Habitat area, quality and connectivity: striking the balance for efficient conservation. J Appl Ecol 48(1):148–152

Ivankova NV, Creswell JW, Stick SL (2006) Using mixed-methods sequential explanatory design: from theory to practice. Field Method 18(1):3–20

Jackson SD, Griffin CR (2000) A strategy for mitigating highway impacts on wildlife. In: Messmer TA, West B (eds) Wildlife and highways: seeking solutions to an ecological and socio-economic dilemma. The Wildlife Society, Bethesda, pp 143–159

Jaspers CJ, Mouissie M, Wessels S, Barke J, Kolen M, Bucholc A (2016) Natuurpunten-systeem voor uniforme waardering van natuurkwaliteit ('Nature points (= T-EQA) system for uniform valuation of nature quality'). Sweco, Houten

Jick TD (1979) Mixing qualitative and quantitative methods: triangulation in action. Admin Sci Q 24(4):602–611

Kormann UG, Scherber C, Tscharntke T, Batáry P, Rösch V (2019) Connectedness of habitat fragments boosts conservation benefits for butterflies, but only in landscapes with little cropland. Landsc Ecol 34(5):1045–1056

Krijn M, Wymenga E (2018) Evaluatie MJPO. Uitwerking Case Studies. A&W-rapport 2473. Altenburg & Wymenga ecologisch onderzoek

Lawton JH, Brotherton PNM, Brown VK, Elphick C, Fitter AH, Forshaw J, Southgate MP (2010) Making space for nature: a review of England's wildlife sites and ecological network. Report to DEFRA, 107

Lesbarreres D, Fahrig L (2012) Measures to reduce population fragmentation by roads: what has worked and how do we know? Trends Ecol Evol 27(7):374–380

Li T, Shilling F, Thorne J, Li F, Schott H, Boynton R, Berry AM (2010) Fragmentation of China's landscape by roads and urban areas. Landsc Ecol 25(6):839–853

Lindenmayer DB, Fischer J (2006) Habitat fragmentation and landscape change. Island Press, Washington

Mimet A, Clauzel C, Foltête JC (2016) Locating wildlife crossings for multispecies connectivity across linear infrastructures. Landsc Ecol 31(9):1955–1973

NDFF, Dutch National Database on Flora and Fauna. www.ndff.nl/english

Ovaskainen O (2012) Strategies for improving biodiversity conservation in the Netherlands: enlarging conservation areas vs. constructing ecological corridors. Report to the Dutch Council for the Environment and Infrastructure, Helsinki

Ovaskainen O (2013) How to develop the nature conservation strategies for the Netherlands? De Levende Natuur 114(2):59–62

PBL & WUR (2017) Lerende evaluatie van het Natuurpact. Naar nieuwe verbindingen tussen natuur, beleid en samenleving. (Learning evaluation of the 'Natuurpact'. Towards new connections between nature, policy and society). PBL, Netherlands Environmental Assessment Agency, The Hague

Pinheiro J, Bates D, DebRoy S, Sarkar D, R Core Team (2018) nlme: linear and nonlinear mixed effect models

Pouwels R, Wamelink GWW, van Adrichem MHC, Jochem R, Wegman RMA, De Knegt B (2017) MetaNatuurplanner v4.0 - Status A: Toepassing voor Evaluatie Natuurpact [MetaNatuurplanner v4.0 – Status A: Application for Evaluation Natuurpact], Wageningen: Wettelijke Onderzoekstaken Natuur & Milieu WOt-technical report 110

R Core Team (2018) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/

Roedenbeck IA, Fahrig L, Findlay CS, Houlahan JE, Jaeger JA, Klar N, Kramer-Schadt S, van der Grift EA (2007) The Rauischholzhausen agenda for road ecology. Ecol Soc 12(1):11

Rytwinski T, Soanes K, Jaeger JA, Fahrig L, Findlay CS, Houlahan J, van der Ree R, van der Grift EA (2016) How effective is road mitigation at reducing road-kill? A meta-analysis. PLoS One 11(11):e0166941

Sawaya MA, Clevenger AP, Kalinowski ST (2013) Demographic connectivity for ursid populations at wildlife crossing structures in Banff National Park. Conserv Biol 27(4):721–730

Sawaya MA, Clevenger AP, Schwartz MK (2019) Demographic fragmentation of a protected wolverine population bisected by a major transportation corridor. Biol Conserv 236:616–625

Sijtsma FJ (2006) Project evaluation, sustainability and accountability: combining cost-benefit analysis (CBA) and multi-criteria analysis (MCA), Dissertation, University of Groningen

Sijtsma FJ, van der Bilt WG, van Hinsberg A, De Knegt B, van der Heide CM, Leneman H, Verburg R (2017) Planning nature in urbanized countries: an analysis of monetary and non-monetary impacts of conservation policy scenarios in the Netherlands. Heliyon 3(e00280):1–30. https://doi.org/10.1016/j.heliyon.2017.e00280

Sijtsma FJ, van der Heide CM, van Hinsberg A (2011) Biodiversity and decision-support: integrating CBA and MCA, Chapter 9. In: Hull A, Alexander E, Khakee A, Woltjer J (eds) Evaluation for participation and sustainability in planning. Routledge, London, pp 197–218

Sijtsma FJ, van der Heide CM, van Hinsberg A (2013) Beyond monetary measurement: how to evaluate projects and policies using the ecosystem services framework. Environ Sci Policy 32:14–25. https://doi.org/10.1016/j.envsci.2012.06.016

Sijtsma FJ, van Hinsberg A, Kruitwagen S, Dietz FJ (2009) Natuureffecten in de MKBAs van projecten voor integrale gebiedsontwikkeling [Ecological effects in MCBAs of projects for integrated area development]. Bilthoven: Netherlands Environmental Assessment Agency. http://www.pbl.nl/nl/publicaties/2009/natuureffecten-in-de-mkba-s-van-projecten-voor-integrale-gebiedsontwikkeling.html

Soanes K, Vesk PA, van der Ree R (2015) Monitoring the use of road-crossing structures by arboreal marsupials: insights gained from motion-triggered cameras and passive integrated transponder (PIT) tags. Wildl Res 42(3):241–256

Strijker D, Sijtsma FJ, Wiersma D (2000) Evaluation of nature conservation. Environ Resour Econ 16(4):363–378

Taylor BD, Goldingay RL (2010) Roads and wildlife: impacts, mitigation and implications for wildlife management in Australia. Wildl Res 37(4):320–331

Torres A, Jaeger JA, Alonso JC (2016) Assessing large-scale wildlife responses to human infrastructure development. Proc Natl Acad Sci 113(30):8472–8477

van der Grift EA (2005) Defragmentation in the Netherlands: a success story? Gaia 14(2):144–147

van der Grift EA (2010) *Richtlijnen voor het meten van het gebruik van faunapassages* [Guidelines for measuring the use of wildlife crossing structures]. MJPO. https://library.wur.nl/WebQuery/wurpubs/419515

van der Grift EA, Dirksen J, Jansman HAH, Kuipers H, Wegman RMA (2009) *Actualisering doelsoorten en doelen Meerjarenprogrmma Ontsnippering* [Actualisation target species and goals Dutch Defragmentation Program] (No. 1941). Alterra

van der Grift EA, Pouwels R (2006) Restoring habitat connectivity across transport corridors: identifying high-priority locations for de-fragmentation with the use of an expert-based model. In: Davenport J, Davenport JL (eds) The ecology of transportation: managing mobility for the environment. Springer, Dordrecht, pp 205–231

van der Grift EA, van der Ree R (2015) Guidelines for evaluating use of wildlife crossing structures. In: Handbook of road ecology, pp 119–128

van der Grift EA, van der Ree R, Fahrig L, Findlay S, Houlahan J, Jaeger JA, Olson L (2013) Evaluating the effectiveness of road mitigation measures. Biodivers Conserv 22(2):425–448

van der Hoek D-J, Smit M, van Broekhoven S, van Hinsberg A, Giesen P, Bredenoord H, Pouwels R, de Knegt B, van Gaalen F, de Blaeij A, Mylius S, Folkert R (2017) Potentiële bijdrage van provinciaal natuurbeleid aan Europese biodiversiteitsdoelen. Achtergrondrapport lerende evaluatie van het Natuurpact (Potential contribution of provincial nature policy to European biodiversity targets. Background report to to the learning evaluation of the Nature Pact), The Hague, PBL (Netherlands Environmental Assessment Agency)

van der Ree R, Heinze D, McCarthy M, Mansergh I (2009) Wildlife tunnel enhances population viability. Ecol Soc 14(2):7

van der Ree R, Jaeger JA, van der Grift EA, Clevenger AP (2011) Effects of roads and traffic on wildlife populations and landscape function: road ecology is moving toward larger scales. Ecol Soc 16(1):48

van Gaalen F, Van Hinsberg A, Franken R, Vonk M, Van Puijenbroek P and Wortelboer R (2014). Natuurpunten: kwantificering van effecten op natuurlijke ecosystemen en biodiversiteit in het Deltaprogramma. (T-EQA quantification of the effects on natural ecosystems and biodiversity in the Delta program). Netherlands Environmental Assessment Agency (PBL) The Hague, 2014. PBL-publication number 1263

van Puijenbroek PJTM, Sijtsma FJ, Wortelboer FG, Ligtvoet W, Maarse M (2015) Towards standardised evaluative measurement of nature impacts: two spatial planning case studies for major Dutch lakes. Environ Sci Pollut R 22(4):2467–2478

van Strien AJ, van Swaay CA, Termaat T (2013) Opportunistic citizen science data of animal species produce reliable estimates of distribution trends if analysed with occupancy models. J Appl Ecol 50(6):1450–1458

Verboom J, Pouwels R (2004) Ecological functioning of ecological networks: a species perspective. In: Ecological networks and greenways: concept, design, implementation. Cambridge University Press, Cambridge, pp 4–72

Wansink DEH, Brandjes GJ, Bekker GJ, Eijkelenboom MJ, van den Hengel B, de Haan MW, Scholma H (2013) Leidraad Faunavoorzieningen bij Infrastructuur [Guideline Fauna Facilities around Infrastructure]. Rijkswaterstaat, Dienst Water, Verkeer en Leefomgeving, Delft/ProRail, Utrecht

Wiertz J, Dirkx GHP, Melman TCP, Reijnen MJSM, Schotman AGM, van Wijk MN, Willemen JPM (2007) *Ecologische evaluatie regelingen voor natuurbeheer: programma beheer en Staatsbosbeheer 2000-2006* [Ecological evaluation for nature management: management program and National State Forest Agency 2000-2006] (No. 500410002, 500410003). Milieu-en Natuurplanbureau

Wilson MC, Chen XY, Corlett RT, Didham RK, Ding P, Holt RD, Holyoak M, Guang Hu, Hughes AC, Jiang L, Laurance WF, Liu J, Pimm SL, Robinson SK, Russo SE, Si X, Wilcove DS, Wu J, Yu M (2016) Habitat fragmentation and biodiversity conservation: key findings and future challenges. Landsc Ecol 31:219–227

Young J, Watt A, Nowicki P, Alard D, Clitherow J, Henle K, Niemela J (2005) Towards sustainable land use: identifying and managing the conflicts between human activities and biodiversity conservation in Europe. Biodivers Conserv 14(7):1641–1661

Publisher's Note Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.