KTH Land and Water
Resources Engineering

Road Ecology for Environmental Assessment

Mårten Karlson

November 2015

TRITA-LWR PHD-2015:06
ISSN 1650-8602
ISBN 978-91-7595-746-3

PhD thesis

Environmental Management and Assessment Research Group

Division of Land and Water Resources Engineering

Department of Sustainable Development, Environmental Science and Engineering

School of Architecture and the Built Environment

Royal Institute of Technology (KTH)

SE-100 44 Stockholm, Sweden

Reference to this publication should be written as: Karlson M, (2015) "Road Ecology for Environmental Assessment". PhD thesis

TRITA-LWR PHD-2015:06

SUMMARY IN SWEDISH

Transportinfrastruktur är nära kopplat till flera politiskt relevanta hållbarhetsfrågor, och en formaliserad miljöbedömningsprocess är sedan 1985 kopplad till planering och byggande av nya vägar och järnvägar i EU (EU-direktiv 85/337/EEG och 2001/42). Syftet med miljöbedömning är att tänka i förväg, att identifiera, förutse och utvärdera eventuella miljöförändringar som kan komma att följa av en föreslagen verksamhet, för att undvika onödiga och oönskade konsekvenser. Biologisk mångfald är en förutsättning för hållbar utveckling som på många sätt påverkas av väg- och järnvägsbyggen, och som har varit utmanande att fullt ut ta hänsyn till inom miljöbedömningsprocessen.

Denna avhandling presenterar fyra studier om biologisk mångfald i miljöbedömning av väg- och järnvägsplaner och projekt. Artikel I sammanfattar aktuell kunskap om vägar och järnvägars påverkan på ekologiska processer, och granskar behandlingen av biologisk mångfald i ett urval av miljöbedömningsrapporter från Sverige och England. Artikel II presenterar en kvantitativ bedömning av det svenska vägnätets påverkan på fåglar och däggdjur, samt en analys av hur habitatfragmentering och vägrelaterade störningar skulle kunna påverka ett urval av ekologiska profiler. Artikel III visar hur vetenskapliga modeller, data och kunskap kan mobiliseras för att utforma och utvärdera järnvägskorridorer, och i Artikel IV analyseras hur landskapets sammansättning tillsammans med storlek och antal faunapassager påverkar habitatkonnektvitet, som en förutsättning för genetiskt utbyte i landskap med betydande barriärer.

Resultaten från Artikel I visar att vägar och järnvägars påverkan på den biologiska mångfalden bör beaktas på alla nivåer av planering; från korridorens dragning i landskapet till vägen eller järnvägens nyttjande och underhåll. Granskningen av miljöbedömningsrapporter visar att hanteringen av påverkan på biologisk mångfald i miljöbedömning har förbättrats under åren, men att brister kvarstår i hur influensområden avgränsas och i hur habitatfragmentering och påverkan på habitatkonnektivitet analyseras.

Resultaten från Artikel II identifierar att naturliga gräsmarker och ädellövsskog, habitattyper som är utpekade som viktiga för den biologiska mångfalden i Sverige, sannolikt påverkas i hög grad av vägspecifika störningar och effekter. Resultat från Artikel II indikerar även att vägrelaterade störningseffekter kan ha en betydande påverkan på den totala tillgängligheten av habitat för ett representativt urval av fåglar och däggdjur. Resultaten från Artikel III demonstrerar hur man genom systematisk kartläggning av ekologiska och geologiska resurser för järnvägskonstruktion kan få dessa att konvergera mot resurseffektiva korridoralternativ med låg påverkan på ekologiska processer. Resultaten från Artikel IV visar att byggandet av ett antal små faunapassager skulle leda till bättre konnektivitet över en barriär än byggandet av en enda stor. Effektiviteten av en fauna passage kommer också till stor del att bero på placeringen av passagen i förhållande till målartens habitatpreferenser.

Denna avhandling visar hur kvantitativa metoder kan bidra till miljöbedömning av biologisk mångfald, och hur påverkan som varit svår att redogöra för i miljöbedömning t.ex. fragmentering och habitatkonnektivitet, kan hanteras. Det rekommenderas att användningen av kvantitativa metoder

och verktyg för prediktion och utvärdering av biologisk mångfald ökar framöver. Framtida utmaningar inkluderar verifiering och kalibrering av befintliga relevanta ekologiska modeller, och ytterligare integrering av vägekologisk kunskap i väg- och järnvägsplanering.

ACKNOWLEDGEMENTS

I would like to express my gratitude towards my supervisor Berit Balfors, how gave me the opportunity to become a PhD-student in the first place, and for her continuous support and guidance. I would like to thank Joanne Fernlund, who initiated the research program GESP, of which my PhD was one part. I would like to thank Ulla Mörtberg who has been deeply involved in my work, for all the energy she has put into reviewing and helping me develop the four research articles in this thesis. Many thanks to Caroline Karlsson and Andreas Seiler for good cooperation with research article three and four. To the department staff and other PhD students I would like to say thanks for nice discussions, fikas, Christmas dinners and other nice events organized. Special thanks Robert Earon who helped review the language of this thesis and to Aira and Britt for helping me out with administrative issues such as travel bills and parental leave reporting. Last but not least, I will acknowledge my family. Thanks to Jannike for providing the necessary space to complete this PhD, and thanks to Sigrid and Ida for being such wonderful kids. Thanks to mom, dad my two sisters and my brother for your enthusiasm, support and feedback. It has been four fantastic years altogether, and I am sure I will remember them as such. The funding for this research project was provided by the Swedish Research Council for Environment, Agricultural Sciences and Spatial Planning, FORMAS.

Mårten Karlson, October 2015

TABLE OF CONTENTS

List of appended papers

Paper I:

Karlson M., Mörtberg U., Balfors B. 2014. Road ecology in environmental impact assessment. *Environmental Impact Assessment Review 48: 10-19.*

Paper II:

Karlson M., Mörtberg U. 2015. A spatial ecological assessment of fragmentation and disturbance effects of the Swedish road network. *Landscape and Urban Planning 134: 53-65.*

Paper III:

Karlson M., Karlsson C.S.J., Mörtberg U., Olofsson B., Balfors B. 2015. Identification and evaluation of railway corridors based on spatial ecological and geological criteria. *Submitted to Transportation Research Part D: Transport and the Environment* 2015-09-29.

Paper IV:

Karlson M., Seiler A., Mörtberg U. 2015. The effect of fauna passages and landscape characteristics on barrier mitigation success. *Submitted to Ecological Engineering 2015-10-13.*

The aims and objectives of the different studies have been formulated in consent with the listed co-authors. Co-authors have contributed with data in Paper II, and data and analysis in Paper III. The author has been exclusively responsible for the execution of the methodologies described in Paper I, II and IV and main responsible for the methodologies described in Paper III. The author is exclusively responsible for the results as well as any graphical representation of the results. Paper I, III, IV were written by the author and reviewed and modified by co-authors. Paper III was written together with the second author, and reviewed and modified by co-authors.

LIST OF ABBREVIATIONS

Abbreviation	Explanation
EIA	Environmental Impact Assessment
SEA	Strategic Environmental Assessment
EIS	Environmental Impact Statement
ER	Environmental Report
GIS	Geographic Information System
MCDA	Multi Criteria Decision Analysis
MSA	Mean Species Abundance
CA	(Habitat) Class Area
NP	Number of (habitat) Patches
PC	Probability of Connectivity
GBC	Generalized Betweenness Centrality
LCP	Least Cost Path
m.a.s.l.	Meters above sea level

Key terminology	Explanation
Patch	A clearly delimited habitat area
Matrix	Areas defined as non-habitat
Habitat suitability map	A classification of a map (e.g. a landcover map) into suitability according to the habitat preferences of a species
Fragmentation	The splitting up of habitat into smaller pieces
Structural Connectivity	Implies that habitat areas area physically connected
Functional connectivity	Implies a possibility of migration between habitat
Barrier effects	Structures or activities impeding movement between habitat

ABSTRACT

Transport infrastructure is closely linked to several politically relevant sustainability issues, and since 1985 a formalized environmental assessment process is linked to planning and construction of new roads and railways in the EU (EU directives 85/337/EEC and 2001/42). The aim of the environmental assessment process is to think in advance; to identify, predict and evaluate significant environmental changes resulting from a proposed activity, in order to adjust the proposed activity accordingly and to avoid unnecessary and unexpected consequences. Biodiversity is a component of sustainable development that is in many ways affected by road and railway construction, but which has been challenging to fully account for within the environmental assessment process. This thesis presents four studies on the role of biodiversity in environmental assessment of road and railway plans and projects. Paper I presents the state of the art of road and railway impacts on ecological patterns and processes sustaining biodiversity, and reviews the treatment of biodiversity in a selection of environmental assessment reports from Sweden and the UK. Paper II presents a quantitative assessment of the impact of the Swedish road network on birds and mammals, and how fragmentation and road disturbance might affect a selection of ecological profiles. Paper III demonstrates how scientific models, data and knowledge can be mobilized for the design and evaluation of railway corridors, and Paper IV analyses how habitat connectivity, as a prerequisite of genetic exchange, relates to landscape composition and size and number of fauna passages. The results from Paper I show that road and railway impacts on biodiversity need to be addressed at every level of planning; from corridor alignment in the landscape to utilization and maintenance. The review of environmental assessment reports shows that the treatment of biodiversity in environmental assessment has improved over the years, but that problems with habitat fragmentation, connectivity and the spatial delimitation of the impact assessment study area remain. The results from Paper II identify natural grasslands and southern broadleaved forest, prioritized habitat types important for biodiversity, to most likely be highly affected by road impacts, and suggest road disturbance to have a high impact on overall habitat availability. The results from Paper III demonstrate how the landscape specific distribution of ecological and geological resources can be accounted for in railway corridor design, and potentially lead to more resource efficient outcomes with less impact on ecological processes. The results from Paper IV indicate that the several small fauna passages would increase connectivity more across a barrier than the construction of a single large. Effective barrier mitigation will also depend on the selection of focal species and the understanding of how the focal species perceive the landscape in terms of resistance to movement. This thesis demonstrates how quantitative assessment can benefit biodiversity impact analysis and address issues such as habitat connectivity and fragmentation, which have been difficult to account for in environmental assessment. It is recommended that biodiversity impact analysis moves towards an increasing use of quantitative methods and tools for prediction, evaluation and sensitivity analysis. Future challenges include verification and calibration of relevant spatial ecological models, and further integration of road ecology knowledge into road and railway planning.

Keywords: Roads, Railways, Biodiversity, Environmental Assessment, GIS, Decision Support

1. INTRODUCTION

Transport infrastructure is closely linked to several sustainability issues of political relevance, i.e. biodiversity and ecological processes, resource and energy efficiency and economic development, all of which need to be addressed in planning. Roads and railways induce changes in ecosystems at multiple scales, from the microclimatic processes in the road or railway corridor to the population dynamics and dispersal possibilities of different species (Coffin, 2007; Davenport and Davenport, 2006; Forman et al., 2003; Trombulak and Frissell, 2000; Spellerberg, 1998). The effects of roads and railways can be measured as the distance from the road or railway, within which changes in species diversity and abundance as well as in hydrological flows, erosion and sedimentation rates can be observed, relative to a control location. This approach to quantification of effects is referred to as the “road effect zone” , and has become a central methodology in road ecology research (Forman and Deblinger, 2000; Forman and Lauren, 1998).

Road and railway environments benefit certain species and might constitute attractive habitat and accelerate ecological processes such as colonization and dispersal (Fahrig and Rytwinski, 2009; Forman et al., 2003). The overall impact from transportation networks on biodiversity is, however, considered to be negative by several authors, who conclude that transport infrastructure has detrimental effects on terrestrial and aquatic communities (Benítez-López et al., 2010; Fahrig and Rytwinski, 2009; Trombulak and Frissell, 2000).

The effects of roads and railways are also reflected in environmental policy. In Sweden and the EU, the construction of transportation infrastructure is to be preceded by an Environmental Impact Assessment (EIA) process according to EU Directive 85/337/EEC concerning projects, and transport infrastructure plans and programs are to be preceded by a Strategic Environmental Assessment (SEA) process according to EU Directive 2001/42. As much as these processes aim to reduce uncertainty on the negative impacts of roads and railways, they also constitute an opportunity to take advantage of the positive changes that accompany road and railway construction. During these processes, impacts are to be identified, predicted and evaluated and concerned stakeholders informed and consulted. The findings of the process are summarized in an environmental impact statement (EIS, for projects) and an environmental report

(ER, for plans), which are used to inform stakeholders and the general public as well as to provide decision support to decision makers (Glasson et al. 2001; Therivel 2004).

Assessing the effects on biodiversity of roads and railways is however a complex task. One major challenge is the variation of responses of different taxonomic groups to the construction and presence of a road or railway. Several plant and insect species, as well as carrion feeding birds and mammals benefit from road and railway environments (Fahrig and Rytwinski, 2009; Lennartsson and Gylje, 2009). By contrast, amphibian and reptile species are often detrimentally impacted by road mortality (Fahrig and Rytwinski, 2009; Eigenbrod et al., 2009; Row et al., 2007; Fahrig et al., 1995), while birds and larger mammals often show declining population densities with increasing proximity to roads (Benítez-López et al., 2010; Helldin and Seiler, 2003; Forman and Deblinger, 2000; Reijnen et al., 1996).

A second major challenge is the delayed response of ecosystems to change and disturbance, frequently observed in studies on the effects of anthropocentric pressures on biodiversity (Folke et al., 2004; Scheffer et al., 2001). Some effects such as habitat fragmentation, traffic noise and change of hydrological, erosion and sedimentation patterns will not immediately result in environmental changes but might gradually and sometimes irreversibly change ecosystem functioning (Eigenbrod et al., 2009; Metzger et al., 2009; Frair et al., 2008; Folke et al., 2004; Findlay and Bourdages, 2000).

A third major challenge is which scale to use for the analysis and assessment of road and railway impacts. At the scale of the road corridor, an environmental assessment might reveal opportunities to improve the conditions for biodiversity, e.g. by the creation of habitat for insects such as pollinators (Ottosson et al., 2012). Assessing fragmentation effects or changes in habitat connectivity would instead require a network approach and a representation of the landscape that matches the spatial distribution of the response variable modelled, as well as the length of the corridor scheme (Beier et al., 2008; Tischendorf and Fahrig, 2000). Assessing road or railway plans and strategies covering vast geographical areas or entire countries would reduce the level of detail possible to analyse, and thereby increase uncertainties. Environmental monitoring would be recommended by the EIA and SEA regulations in such cases.

The efficiency of environmental assessment in meeting the intended sustainability objectives has been thoroughly discussed in recent years, and concerning EIA, it has been argued that there is a gap between what is required by the legislation and what is carried out in practice concerning (Morrison-Saunders and Retief, 2012). A number of articles that have specifically criticized on the treatment of biodiversity issues in EIA and SEA of roads and railways indicate that this gap between legislation and practice concerns planning and construction of transport infrastructure as well (Karlson et al., 2014; Gontier et al., 2006; Byron et al., 2000; Thompson et al., 1997).

Although recent research suggested that there are substantial improvements regarding the treatment of biodiversity in environmental assessment of transport infrastructure, some problems seem to persist (Karlson et al., 2014). Recent reviews of EIA and SEA practice concluded that analyses of road and railway effects on biodiversity were predominantly carried out through mapping of biodiversity features and consultation of experts, stakeholders and the public (Karlson et al., 2014; Söderman, 2009, 2006). The environmental assessments were found to be descriptive rather than analytical and predictive, and in the final reports summarizing the EIA or SEA process, impacts on biodiversity were communicated through descriptive texts and qualitative evaluation matrices. It was further argued in Karlson et al. (2014) that the concurrent lack of use of quantitative methods, measurable indices and predictive tools may reduce the amount of biodiversity issues that can be addressed during an environmental assessment process, and that it omitted analyses of certain important impacts such as fragmentation effects and changes in habitat connectivity.

Roads and railways today cover approximately 1.2 % of the Swedish land area (SCB, 2013), and the Swedish Transport Agency was recently provided with 54.8 bilj. EUR in funding for maintenance and capacity increasing actions for the years 2014-2025 (Ministry of Enterprise and Innovation, 2014). Such large investments could provide opportunities to substantially improve the environmental performance of the existing transportation network, as well as to let future planning and design be informed by road ecology (e.g. (van der Ree et al., 2015; Forman et al., 2003), in order to avoid unnecessary negative effects.

This thesis attempts to review and develop methods for reducing the gap between scientific and practical environmental assessment of roads and railways, and presents four studies on the role of biodiversity in road and railway planning. The first study presents a summary of the main impacts that roads and railways have on ecological processes, and reviews the state of the treatment of these impacts in environmental assessment documents from a selection of EIA and SEA processes. The second study presents the "Swedish case", by modelling and visualization of road effects published in the scientific literature, using the national road network and a set of response variables representative for a selection of Swedish birds, mammals and valuable nature types. The third study demonstrates a railway corridor planning model, in which alternative railway corridors were created and evaluated through integrated modelling of ecological and geological criteria for corridor localization and design. The fourth and final study concern the barrier effect of roads and railways, and analyse different scenarios of fauna passages and landscape composition.

1.1. Aim and objectives

The overall aim of this thesis was to contribute to road ecology research with knowledge on its application in environmental assessment and with the development of methods that would facilitate the integration of biodiversity aspects in environmental assessment of roads and railways. Specific objectives of the thesis were: 1) to review the scientific literature on road and railway effects on biodiversity in order to relate biodiversity impacts to specific phases of the transport infrastructure life cycle; 2) to review the treatment of biodiversity in road and railway environmental impact statements and reports in order to comprehend the contemporary level of integration of road ecology in environmental assessment (Paper I); 3) to estimate and assess the overall impact of the national road network, and to compare the potential effects of fragmentation and disturbance on selected groups of birds and mammals in Sweden (Paper II); 4) to develop methods for the integration of ecological and geological sustainability criteria for railway corridor planning, and to demonstrate how to quantify and evaluate the performance of different railway corridors (Paper III); and 5) to analyse the effects of different configurations of fauna passages and landscape composition on connectivity as a prerequisite to genetic exchange between populations (Paper IV).

1.2. Structure of the thesis

An introduction to concepts and theories that are relevant for the research presented in this thesis is given in Chapter 2. Chapter 3 describes the methods used in the four different studies. Chapter 4 summarises the state of the art for road ecology as a research area, for the treatment of biodiversity in environmental assessment and the results of the four studies. The results of the different studies, together with future challenges and opportunities for road ecology and environmental assessment research are discussed in Chapter 5. Concluding remarks and ideas for future research are presented in Chapter 6 and 7.

2. Theoretical background

2.1. Road ecology

Road ecology is the study of environmental changes related to roads and vehicles specifically, but in a broader sense relevant for linear transportation systems in general (Davenport and Davenport, 2006; Forman et al., 2003). Road ecology is based both on basic research disciplines and methods such as wildlife ecology and hypothesis testing, on applied sciences such as conservation biology and environmental engineering, and on social science research disciplines studying the infrastructure planning process and the effective implementation of scientific conclusions and recommendations. Central to road ecology is the study of impacts of roads on ecological processes, including effect zones, habitat fragmentation and connectivity and animal mortality. The “road effect zone”, first coined by Forman (1998), is the distance

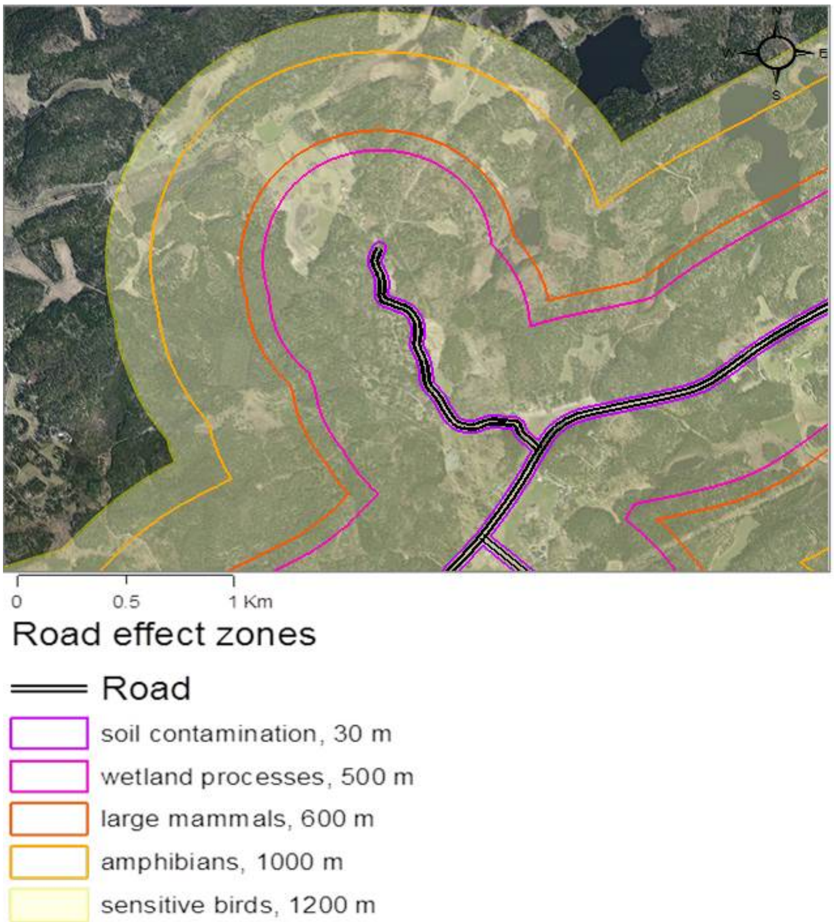

Figure 1. Illustration of road effect zones based on empirical studies (Eigenbrod et al., 2009; Gagnon et al., 2007; Bäckström et al., 2003; Forman et al., 2002; Findlay and Houlahan, 1997).

perpendicular to any point at a road within which environmental changes can be significantly distinguished from a control location (Fig. 1). While the measuring of effect zones is a methodology developed within road ecology, the study of fragmentation and habitat connectivity is based on methods developed within landscape ecology and meta-population theory. Landscape ecology is the study of how landscape patterns, i.e. the distribution of biotic and abiotic elements, influence ecological (including anthropogenic) processes and vice versa (Bastian et al., 2002; Forman and Godron, 1986). Meta-population research is the study of which factors influence the persistence of species populations in terms of colonization and extinction. A meta-population is a network of sub-populations of a species, between which there are certain rates of migration of individuals (Fig. 2). Meta-population theory links the population level processes mortality and reproduction to landscape patterns such as habitat amount and distribution, and provides models for how these determine the rate of population colonization and extinction in a network of suitable habitat areas (Hanski, 1994; Hanski and Gilpin, 1991).

Crucial for the modelling of meta-population dynamics, inter alia the persistence of species populations in fragmented landscapes is an accurate representation of species migration patterns in the landscape. A prerequisite for a successful migration between two populations is a minimum degree of functional connectivity, i.e. populations do not have to be structurally connected but migratory individuals must be able to successfully reach their destination and produce offspring at some point in their lifetime (Hanski and Ovaskainen, 2003; Hanski, 1994). Metrics for species movement and dispersal between populations commonly use geographical distance as predictor for migration success, sometimes weighted by some factor of importance, e.g. probabilities, observations, genetic differentiation, maximum movement capacity, among others (for a good overview, see e.g. Etherington and Holland(2013), Pinto and Keitt (2009)). Connectivity can be represented as a function of the landscapes resistance to movement, and modeled as electrical flows using Ohms' law (Shah and McRae, 2008; McRae et al., 2008; McRae, 2006). A landscape represented by a raster map would then constitute a network of resistors, in which pixels "become" resistors according to a user defined rule. Current strength, inter alia connectivity, as a function of resistance can then be calculated at any point in the map, making possible the identification of areas with high current densities and thus landscape structures receiving high flows. Such information would be useful for planning and design of new linear infrastructure, as well as for mitigation activities aiming towards reducing negative effects on wildlife migration patterns.

2.2. Environmental assessment

The EU Directive 85/337 on the assessment of the effects of

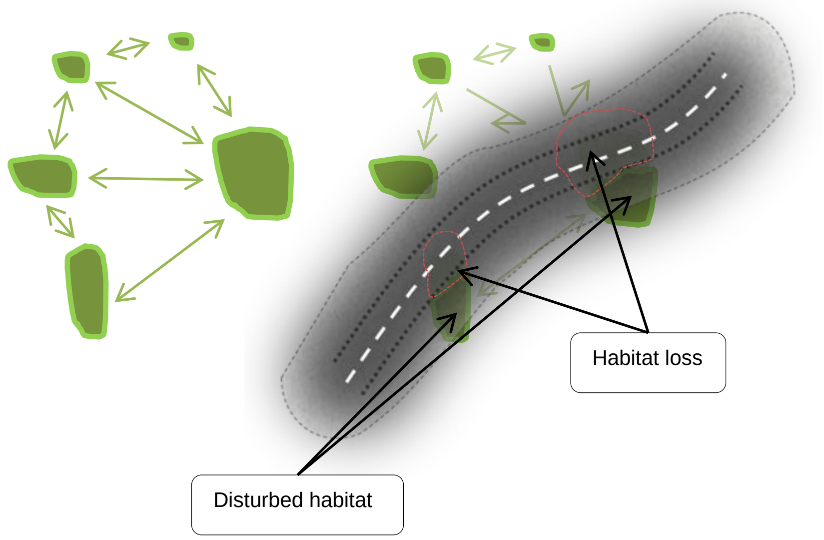

Figure 2. Conceptual illustration of a meta-population, and how the construction of a road might interfere with connectivity between populations, habitat amount and quality.

certain public and private projects on the environment (EIA), and EU Directive 2001/42 on the assessment of the effects of certain plans and programmes on the environment (SEA), are two legislative tools that are developed to prevent adverse environmental impacts and promote sustainable development as defined in the Brundtland report in 1987 (Glasson et al., 2001). The Directives require certain types of plans and projects to be preceded by a process in which the environmental impacts of an activity are to be identified, assessed and evaluated, and stakeholders affected by the activity to be informed and consulted. The processes are then summarized in a report, commonly used as a basis for decision-making and to inform the general public. EIA was first incorporated in the US National Environmental Policy Act of 1969, which led to the development of EIA legislation throughout the world. In 1985 EIA became EU law, and EU Directive 85/337/EEC has since been amended three times: in 1997 to better address trans-boundary impacts (97/11), in 2003 in which the role of the public was strengthened (2003/35) and in 2009, where projects related to transport, capture and storage of carbon dioxide were added to Annex 1 and 2. In the end of 2011, the EIA Directive and its three amendments were codified into directive 2011/92/EU, in which the role of public participation, the presentation of alternative project designs and potential transboundary impacts were emphasized. The need to assess the environmental impacts of plans as well as projects was realized already in 1975, and SEA was more or less required for specific activities in some countries. Environmental assessment of plans as well as projects was originally thought to be regulated by one single directive, but at the time of introduction of the EIA Directive in 1985, only project activities were supported. Further, it was realized that EIA had little influence on project design and location, as there was seldom room for making changes to an activity in its project phase. This emphasized the need for environmental assessment already in the planning phase, and the SEA Directive was launched in 2001 (Therivel, 2004). The EIA and SEA Directives are essentially instruments of sustainable development with the overall aim to prevent rather than repair unintended environmental damage. The research presented in this thesis focuses on how the requirements of the EIA and SEA Directives consider biodiversity aspects in projects and plans, as well as on development of methods for improved integration of these aspects.

2.3. Multi Criteria Decision Analysis

Multi-criteria decision analysis (MCDA) is a family of non-monetary evaluation techniques developed to work with mixed input data types (Munda et al., 1995, 1994). This makes MCDA techniques appropriate for developing decision support on environment-development conflicts, e.g. road and railway planning, as such situations require disparate and incommensurable concerns often measured using different scales, to be integrated and accounted for in planning and decision making processes (see e.g. (Bagli et al., 2011; Huang et al., 2011; Geneletti, 2004). MCDA techniques essentially provide a model and a set of rules for the evaluation of potential outcomes of a decision problem. It includes standardization, ranking and aggregation of the input variables into a comprehensible unit of performance, e.g. between 0 and 1 (Huang et al., 2011; Malczewski, 1999). MCDA techniques are commonly applied in some countries, and have been increasingly applied in environmental sciences the last decade (Huang et al., 2011; Kiker et al., 2005; Janssen, 2001).

3. METHODS

3.1. Literature review (Paper I)

A review of scientific literature was conducted on the topic of ecological impacts of transport infrastructure. Apart from articles reviewing the topics of road ecology and environmental impacts of linear constructions in general, central to this literature review were the topics of ecosystem processes, habitat fragmentation, road mortality and effects resulting from the utilization and maintenance of transportation infrastructure. This review was carried out with the purpose of providing an overview of the ecological impacts of transport infrastructure, and to structure the information according to environmental assessment terminology. Therefore, interactions between transport infrastructure and ecological processes were structured in a table as Action–Effect–Impact. "Action" described the attributes of transport infrastructure that induce changes in the surrounding environment. "Effect" was distinguished from "Impact" using the definition in Catlow and Thirlwall (1976) where "Effect" is the direct or indirect biophysical changes of an activity, and "Impact" was the resulting consequences of those changes.

3.2. Review of environmental reports (Paper I)

Articles reviewing the treatment of biodiversity in environmental assessment of roads and railways were identified in the scientific literature, and a content analysis (Kvale and Brinkman, 2009) was employed to create a list of frequently remarked upon problems. The list of problems were then used as a checklist for review of altogether seven Environmental Reports (ER) and 16 Environmental Impact Statements (EIS) on road or railway projects, plans or programmes produced between 2005 and 2013 in Sweden and the UK (Karlson et al., 2014). The countries Sweden and the UK were selected because they are both subjected to the EIA and SEA Directives, and because these reports were written in English or Swedish. The EIS/ERs where scrutinized for problems with the treatment of ecological impacts, as described by the checklist, and the problems per impact category were quantified.

3.3. Scalable ecological indices (Paper II, III)

Ecological processes occur simultaneously at multiple scales in time and space, which in order to assess relevant potential impacts of a project requires biodiversity indices to be either scale-independent or scalable to the appropriate environmental assessment context. One way of managing the scales issue in biodiversity impact assessment is to construct ecological profiles (Angelstam et al., 2004; Vos et al., 2001; Mörtberg et al. 2012). An ecological profile represents multiple species or multiple groups of species with similar traits, such as habitat requirements and movement capacity, and which therefore can be expected to respond to environmental changes in similar ways (Fig. 3). The concept is similar to that of umbrella species, and follows the same recommendations on selection of model species; high degree of habitat specialization, high sensitivity to disturbance and area demands representative for the scale of context (Edman et al., 2011; Angelstam et al., 2004; Fleishman et al., 2000). As the concept of ecological profiles aims to model the behaviour of a larger system through the behaviour of prioritised biodiversity components, it is imperative to ensure that the formulation of species requirements and attributes is realistic and relevant for the specific context.

3.4. Assessment of road effects (Paper II)

3.4.1. Study area

In order to assess the overall effects of the Swedish road network on relevant biodiversity components, the potential changes in Mean Species Abundance (MSA) of birds and mammals were modelled. In the first step, generic effects, measured as declining Mean Species Abundance (MSA) was modelled using the national road network and the entire Sweden as a study area (Study are A, Fig. 4). The second step consisted of a comparison of the effects

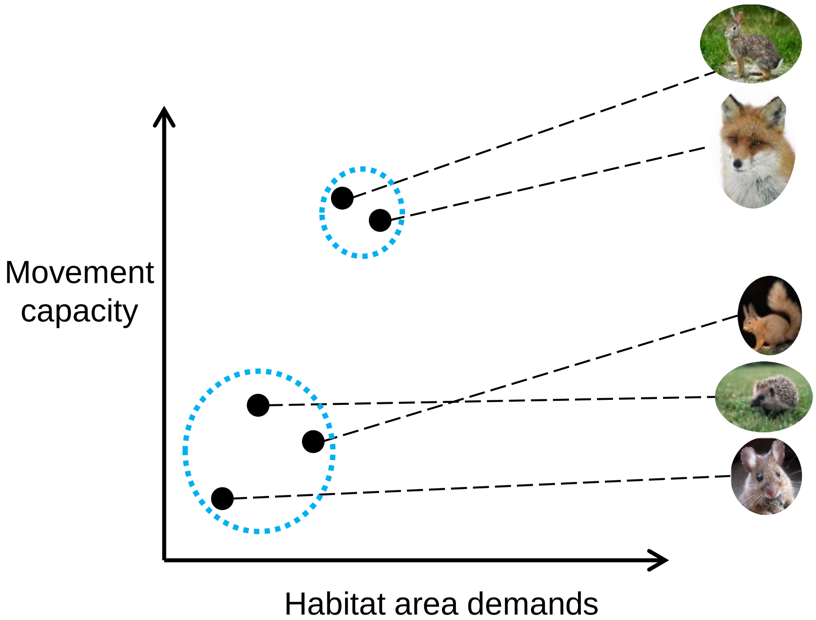

Figure 3. Conceptual model of the ecological profile system. A single species might be chosen to represent others, if they share, to a minimum degree, a selection of traits and resource requirements.

of fragmentation and disturbance on habitat networks for a selection of ecological profiles. A smaller study area located in south-eastern Sweden was used for this purpose (Study Area B, Fig. 4). Study Area B comprised three counties (Stockholm, Uppsala and Västmanland) in their entirety and parts of another four counties (Södermanland, Örebro, Dalarna and Gävleborg) covering a total of 43,827 km^2.

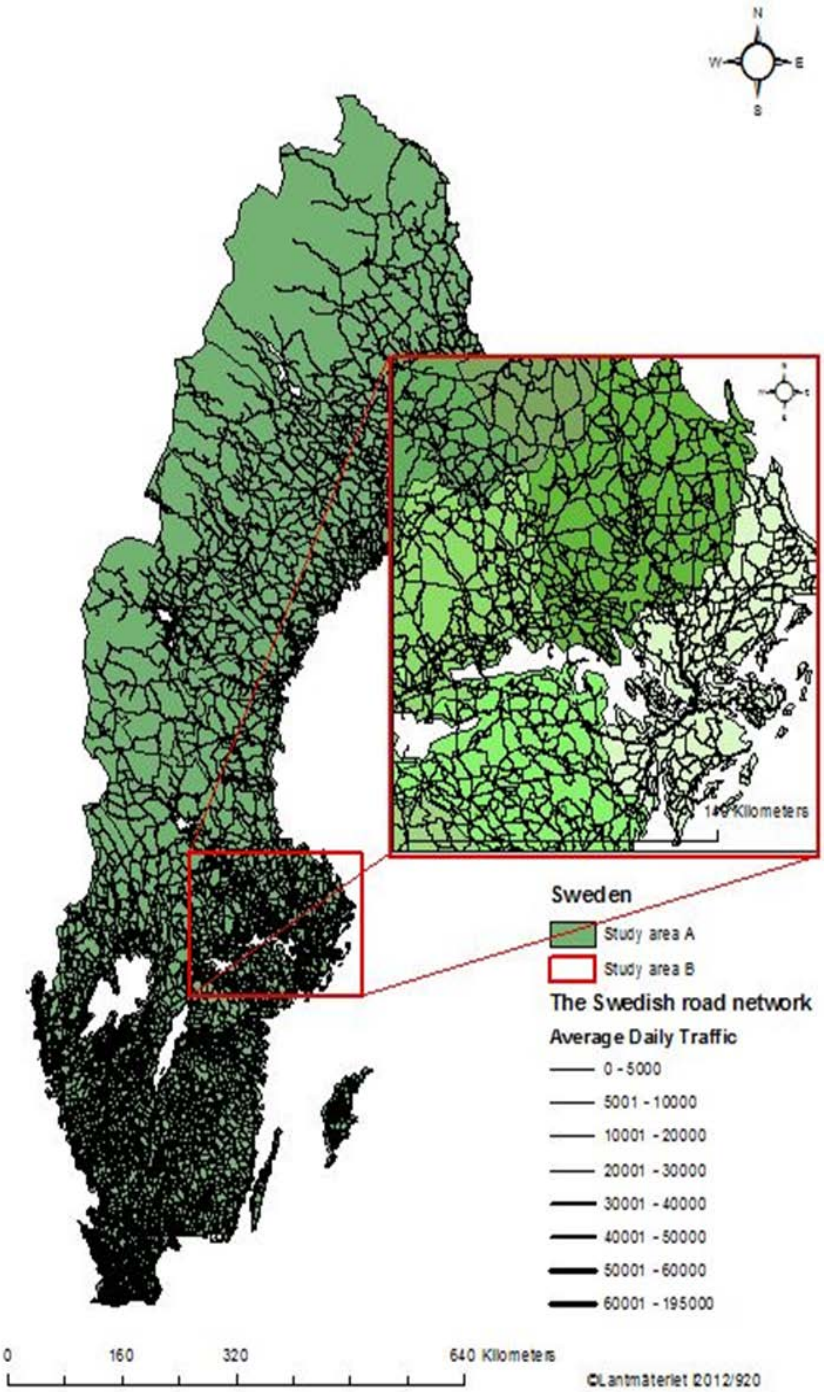

Figure 4. Study area A and B used for the analyses in Paper II.

3.4.2. Effects of the Swedish road network on birds and mammals

For both parts of the study, the spatial distribution of road effects of different intensity was modelled in order to enable analysis and impact assessment. The two selected models published by Benítez-López et al. (2010) on the effects of distance from transport infrastructure on mammals and birds were used to model road effects of the Swedish road network (Fig. 5).

The models show a binomial prediction of the Mean Species Abundance (MSA) of birds and mammals, as a function of distance from transport infrastructure, derived from generalized linear mixed effect models and a logit link function. From these models, two spatial datasets on the predicted road effects on mammals and birds ($MSA_{mammals}$ and MSA_{birds}) and four spatial datasets showing the associated upper and lower 95 % Confidence Intervals (CI) were derived in ArcGIS (ESRI, 2013) at a resolution of 25 m × 25 m. This was done by applying a logit transformation:

$$MSA_{(estimated)} = \frac{e^u}{1 + e^u}$$

where $MSA_{(estimated)}$ was the predicted MSA at the observed distance from the road ranging from 0 to 1, and u was the linear equation describing the log-transformed probability of the presence of a species at a certain distance x from the road:

$$u = \ln\left(\frac{P_i}{1 - P_i}\right) = a + bx$$

where a was the estimated value of u when $x = 0$ and b was the regression coefficient for the independent variable x. The regression coefficient b and the predictions (P_i) were retrieved from supplementary materials from Benítez-López et al. (2010). The distance variable x could take the value of each cell in a raster containing the Euclidian distance from a road, calculated from the national road database (Swedish Transport Administration, STA, 2012). For further inquiry about how effect distances were calculated, see Benítez-López et al. (2010) and Paper II of this thesis.

The derived MSA datasets were then reclassified into four effect intensity zones with break values of 0.5, 0.7 and 0.9, and overlaid on five national scale data sets (Study area A), showing the distribution of nature types with high importance for biodiversity (SEPA, 2012). A generic effect of roads on birds and mammals (represented by the five habitat types used) was then estimated as

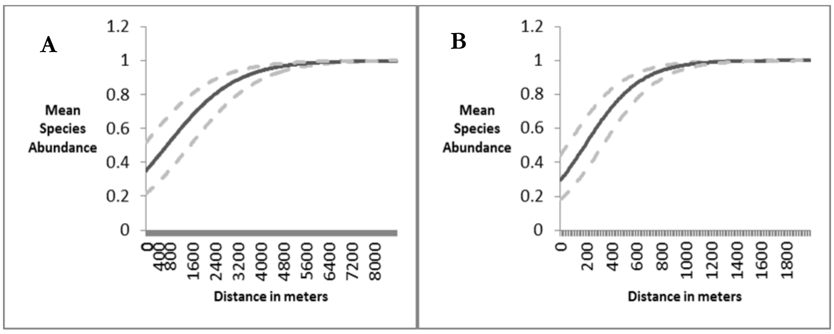

Figure 5. Predictions of the Mean Species Abundance of mammals (A) and birds (B) as an effect of distance to infrastructure, based on statistical analyses of empirical data. Adapted from Benítez López et al., 2010.

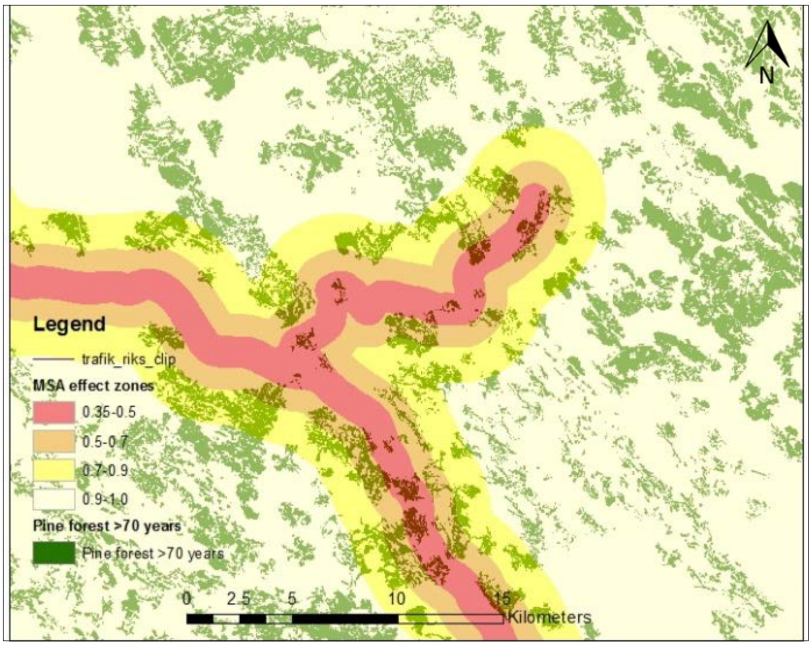

Figure 6. Illustration of the different road effect intensity intervals relative to the distribution of pine forest older than 70 years in a Swedish landscape. Spatial data from SEPA (2012).

the areal intersection between each nature type and the four effect intensity zones. The intersection illustrated the proportion of different habitat types under different level of disturbance from the road network (Fig 6).

3.4.3. Comparing fragmentation and disturbance effects

For the second part of the study (Study area B), the two MSA datasets were reclassified into two datasets with an effect zone of MSA 0.5, to represent depressed species densities as a result from disturbance effects. This dataset was overlaid on habitat suitability maps, showing the spatial distribution of habitat for a selection of ecological profiles (concept explained in Sub-section 3.3). The six ecological profiles represented different forest and grassland dwelling mammals and birds in Sweden, with area demands from approximately 10 ha to 600 ha. The suitability of the habitat maps were reclassified according to the disturbance introduced by the MSA effect-zone layers, and through a modelling framework (explained in Paper II) in which the habitat suitability maps were aggregated into habitat patches, "disturbed" habitat networks of the ecological profiles were derived. "Fragmented" habitat networks were created by overlaying the road network on the original habitat suitability maps and removing pixels that were crossed by the road, before the aggregation into habitat patches. Control habitat networks for the ecological profiles were also created from the habitat suitability maps. The spatial properties of the "disturbed", "fragmented" and the control habitat networks were then quantified in terms of total habitat area (CA) and number of habitat patches (NP), and the different effects could then be assessed by comparing metric scores with those of the control habitat networks.

3.5. Ecological criteria for railway corridor planning (Paper III)

3.5.1. Study area

The study area was located in the Stockholm County, Sweden, in a peri-urban area about 20 km north of Stockholm City (Fig. 7). The area is delimited to the west by the major highway E4 and to the east by a railway used for commuting. Suburban areas with a higher density of infrastructure, residential and commercial areas are situated to the south of the study area. North of the study area, the landscape is relatively open with a mix of forest and agricultural land, intersected by roads with average daily traffic volumes of a few hundred vehicles (STA, 2012). The railway planning proposition analysed in this study (Håkansson et al., 2012) suggested an additional link between Arlanda Airport, the largest international airport in Sweden, and the existing railway system, either at the station Vallentuna or at Lindholmen (Fig. 7).

3.5.2. Indices of habitat connectivity

The Probability of Connectivity (PC) metric (Saura and Pascual-Hortal, 2007) and a generalized, ecologically adapted version of the Betweenness Centrality (GBC) metric (Bodin and Saura, 2010) were used to quantify habitat connectivity. The PC model quantified the contribution to habitat availability of individual habitat patches, relative to a total measure of availability of a larger network. This allowed for individual habitat patches to be ranked accordingly, providing an estimate of patch importance in terms of both size and reachability from other patches. The GBC quantified the centrality of a patch in a network, which provided insight on which patches in the intact habitat network, were likely to function as stepping stones and receive high flows of migrating individuals. The mathematical formulation of the metrics follows as:

$$PC = \frac{\sum_{i=1}^{n}\sum_{j=1}^{n} a_i a_j p_{ij}^*}{A_L^2}$$

where a_i; a_j = the attribute (area in this study) of patch *i* and patch *j* and A_L = the total area of the study area. p_{ij}^* = the maximum product of probability-weighted dispersal pathways between patches *i* and *j*. The calculation of p_{ij}^* allowed for patches to constitute intermediate steps in a dispersal event, if the product probability of dispersal between *i* and *h* via *j* would be higher than a direct dispersal between *i* and *h*, which adds to the ecological realism of this metric.

Patch importance was given by dPC_k, calculated as percent loss of connectivity following a removal event, given by the equation

$$dPC_k = 100 * \frac{(PC - PC_{remove,k})}{PC}$$

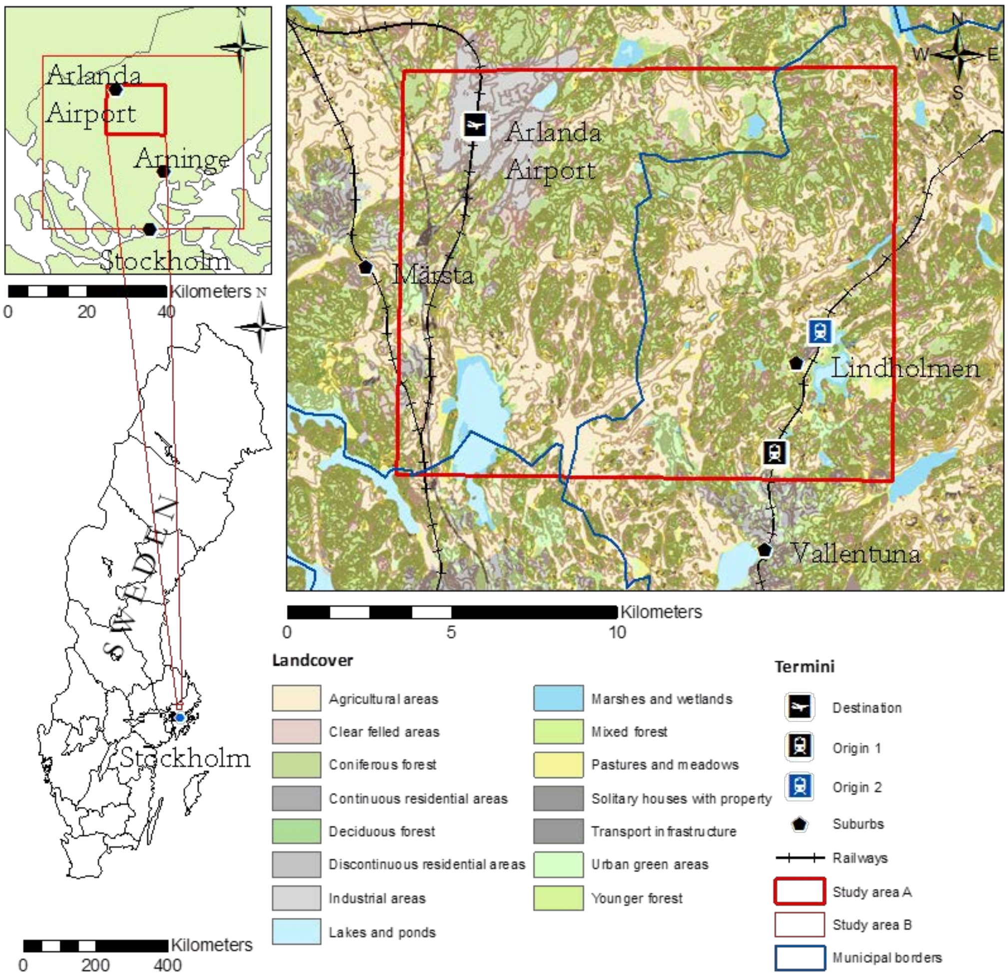

Figure 7. Study area used in Paper III.

The GBC metric is a modification of the original BC metric (Freeman, 1979) through the integration of a fraction of the PC metric, that quantifies habitat availability in terms of network connectivity (Saura and Pascual-Hortal, 2007; Saura and Rubio, 2010). It is estimated by the equation:

$$BC_k^{PC} = \sum_i \sum_j a_i a_j p_{ij}^{*k} \{i, j \neq k, ij \in nm^*\}$$

where k is a given patch in the habitat network and nm^* the list of all possible combinations of patches i and j. a_i , a_j and p_{ij}^{*k} are denoted the same way as in the PC metric.

In these models of habitat connectivity, the distribution of habitat is represented as a patch-matrix landscape where habitat patches are identified as nodes, and movement or dispersal between them as edges, or links, between patches (Fig. 8). The matrix is considered inhospitable, and its contribution (positive or negative) to habitat connectivity is ignored in the calculation of these metrics. Both metrics need to be parameterized with data on the delimitation of habitat and with a measure of movement potential of the model species. In the study (Paper III), expert models on

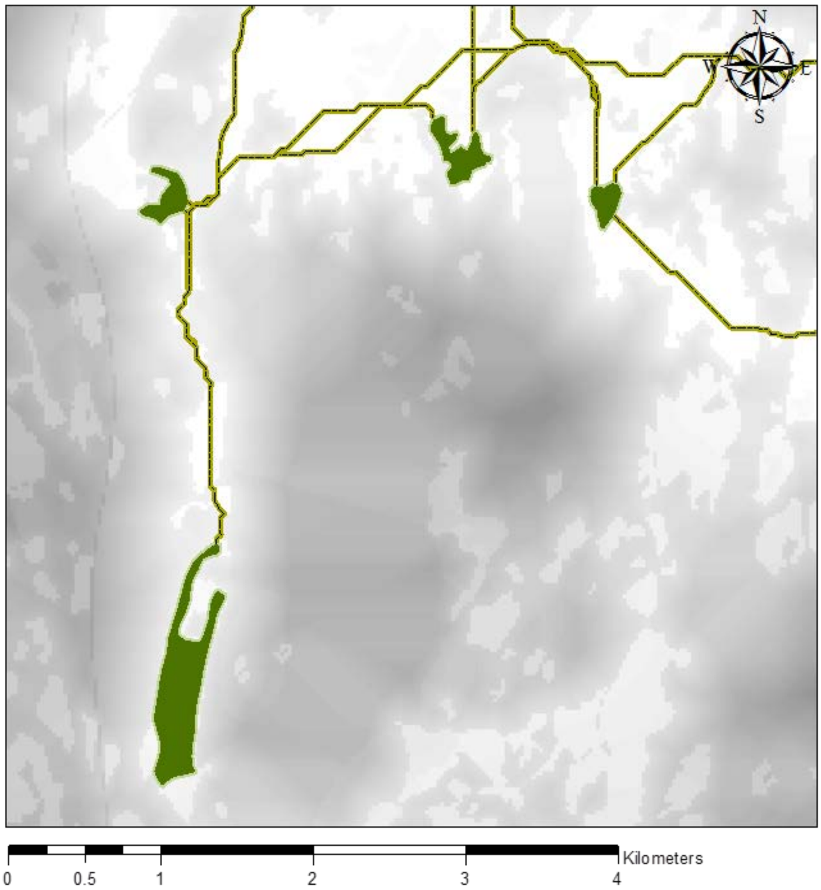

Figure 8. Illustration of habitat patches and movement pathways derived from spatial ecological models employed in Paper III.

habitat preferences and movement capacity were used to define assumptions on habitat areas and dispersal potential for a selection of ecological profiles (see Section 3.3, Paper III).

3.5.3. Railway corridor design and evaluation

The overall modelling framework for corridor planning developed for the study (Paper III) involved a design part and an evaluation part. The design part was carried out in two main steps. The first step consisted of modelling the spatial distribution of important ecological and geological criteria for sustainable railway construction in the study area. The output of these models was standardized and aggregated through the use of GIS and multi-criteria analysis techniques, into railway corridor suitability maps showing areas with high and low suitability for railway construction respectively. Three different suitability maps were created to reflect contrasting perspectives on important criteria for railway construction: one suitability map according to an "ecological perspective", in which ecological criteria were given weight over geological criteria, one according to a "geological perspective", created with the reversed set of weights

Table 1. Decision rules for the SMCA.

<table>
<tr><td>Objective 1</td><td colspan="2">Ecological railway suitability</td><td colspan="2">Geological railway suitability</td></tr>
<tr><td>Constraints</td><td colspan="4">Water bodies with 100 m buffer zones and protected areas</td></tr>
<tr><td rowspan="5">Factors</td><td colspan="2">Weights</td><td colspan="2">Weights</td></tr>
<tr><td rowspan="2">Valuable areas</td><td rowspan="2">0.2</td><td>Geological suitability for construction</td><td>0.35</td></tr>
<tr><td>Geological suitability for aggregates</td><td>0.3</td></tr>
<tr><td>Movement pathways</td><td>0.35</td><td>Slope</td><td>0.1</td></tr>
<tr><td>Stepping stones</td><td>0.45</td><td>Soil thickness</td><td>0.25</td></tr>
<tr><td>Objective 2</td><td colspan="4">Railway corridor suitability</td></tr>
<tr><td>Constraints</td><td colspan="4">Water bodies with 100 m buffer zones and protected areas</td></tr>
<tr><td>Factors</td><td colspan="2">Ecological railway suitability</td><td colspan="2">Geological railway suitability</td></tr>
<tr><td colspan="5">Weights for the different perspectives</td></tr>
<tr><td>Ecological</td><td colspan="2">0.75</td><td colspan="2">0.25</td></tr>
<tr><td>Geological</td><td colspan="2">0.25</td><td colspan="2">0.75</td></tr>
<tr><td>Neutral</td><td colspan="2">0.5</td><td colspan="2">0.5</td></tr>
</table>

and one "neutral" suitability map using an equal set of weights (Table 1). ArcGIS 10.2 (ESRI, 2013), was used to process the results from spatial ecological and geological models, while standardization, weighting and aggregation were performed in IDRISI Andes (Eastman, 2006).

In the second step, the suitability maps were fed into a Least Cost Path (LCP) model for the identification of a total of six railway corridors from two different locations (Vallentuna and Lindholmen) to one common destination (Arlanda Airport). The LCP-model identified the least costly path between two destinations on a raster map, in which each pixel represented a unit of travel cost. The LCP-modelling was executed using ArcGIS 10.2 (ESRI, 2013).

The evaluation part of the study involved three main steps. First, the performance of each corridor was evaluated in terms of mass-balance performance, length and by the ecological criteria habitat loss, broken links and accumulated connectivity loss. Second, the evaluation results for each alternative railway corridor were input to a non-spatial MCA, with the objective to arrive at a preferred corridor. Again, the evaluation was performed according to three sets of weights, reflecting an ecological, a geological and a neutral perspective on corridor performance (Table 2). The final step was

Table 2. Decision rules for the non-spatial MCA. The standardization method linear value function and the aggregation rule additive value function (weighted sum) was used for all factors.

Objective		Identification of the most sustainable railway corridor				
Factors		Habitat loss	Broken links	Accumulated connectivity loss	Mass-balance performance	Corridor length
Weights (direct weighting)	Ecological perspective	37.5	17.5	20	12.5	12.5
	Geological perspective	9	8	8	25	50
	Equal perspective	25	12.5	12.5	25	25

to perform a sensitivity analysis in order to evaluate the robustness of the MCA results.

3.6. The effects of fauna passages and landscapes characteristics on barrier mitigation success (Paper IV)

3.6.1. Hypotheses

Three hypotheses were formulated on the relationship between connectivity, landscape composition and number of fauna passages. Hypothesis one was formulated as (h:1) "Several small passages evenly distributed along a barrier will increase connectivity more than a single large passage"; (h:2) "Increasing contrast will either increase or decrease connectivity depending on if a fauna passage would be located in a highly connected or poorly connected area"; and (h:3) "Increasing landscape aggregation will have an effect on connectivity in scenarios with high contrast, but not in scenarios with low contrast".

3.6.2. Construction of scenarios

Landscape composition was generalized into measures of habitat heterogeneity, aggregation and contrast between habitat types. The assumptions included that there was a constant level of heterogeneity; that any landscape consisted of nine different classes of movement resistance; that landscape aggregation would determine the "patchiness" of a landscape; and that the contrast would determine the disparity between adjacent resistance values (Fig. 9). A landscape was created from a random raster with a uniform distribution of resistance values in a GIS. From this landscape, five new landscapes were created, each with a higher level of aggregation. These six landscapes were then duplicated and exponentially reclassified, to increase contrast between neighbouring pixel values. The landscapes with six levels of aggregation and two levels of contrast were then combined with seven scenarios of barriers and fauna passages. The landscape creation process was iterated over 30 random raster datasets, resulting in 2520 landscapes with different combinations of

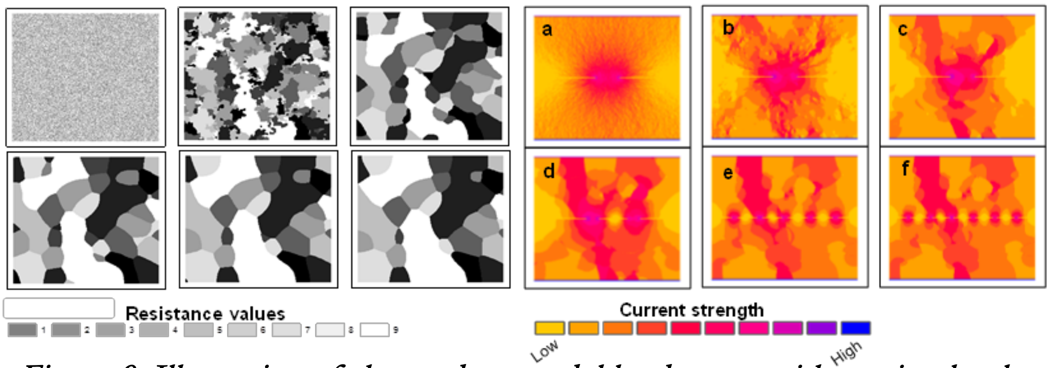

Figure 9. Illustration of the random model landscapes with varying level of aggregation and contrast, created in Paper IV (to the left). To the right we see analysis results from different scenarios of fauna passages and aggregation level. Scenario a) consisted of a zero-aggregated landscape with a uniform distribution of habitat classes and a barrier with one gap, which was assumed to be able to funnel through animals within 20 pixels on either side of the fauna passage, b) shows a scenario with one level of aggregation and one fauna passage, c) shows two levels of aggregation and one fauna passage, d) shows three levels of aggregation and two fauna passages, e) shows five levels of aggregation and five fauna passages, and f) shows five levels of aggregation and six fauna passages.

aggregation, contrast and fauna passages for which connectivity was quantified using circuit theory and related methods.

3.6.3. Habitat connectivity modelling and analysis

Habitat connectivity was modelled using circuit theory, and carried out with the software Circuitscape (Shah and McRae, 2008) (Figures 9a-f). A detailed description of the analysis procedure is provided in Paper IV. Each of the 2520 combinations were analysed in Circuitscape, and the mean and standard deviation (SD) of the 30 iterations were calculated for every pixel in the resulting current maps (resulting in 84 mean current maps and 84 SD current maps). Finally, the sum of mean current and their respective SD was retrieved for each map, and prepared for statistical analysis.

Exploratory analyses including linear regression revealed that assumptions on normality and homogeneity of variances were not met (Shaphiro-Wilks $p > 0.05$, Levene's test $p > 0.05$). So it was decided to use the non-parametric Kruskal Wallis and Man-Whitney U-test for hypothesis testing and post hoc analysis ($\alpha = 0.05$).

The effect of several small versus a single large fauna passage on connectivity (h:1 SLOSS) was initially analysed by multiple linear regression with F-test ($\alpha = 0.05$). The data on mean current from the different scenarios was regressed against number of fauna passages (1-7) and (h:3) level of aggregation (1-6), ($n = 42, df = 2$). The data was then structured according to the SLOSS scenarios into seven samples, and the null hypothesis that there were no differences between samples was contested with Kruskal-Wallis test, followed by Mann-Whitney U-test between samples ($\alpha = 0.05$).

The effect of contrast on connectivity (h:2) was tested for significance ($\alpha = 0.05$) using a Mann-Whitney U-test between the mean current strength from the aggregated and the contrasted combinations as well as between their respective SD. As it was assumed that connectivity could either increase or decrease, a two-tailed test was used.

4. RESULTS

4.1. State of the art – road ecology

4.1.1. Road and railway effects on the environment

From an ecological perspective, roads and railways introduce changes in ecosystems, not only causing local habitat loss, but also disturbances and changes in habitat amount, quality and connectivity and alteration of hydrological processes (Coffin, 2007; Forman et al., 2003). These landscape scale changes impact on ecological processes such as colonization and extinction, species distribution and genetic exchange with indirect impacts on trophic structures and the viability of populations at various scales, and consequently on biodiversity (Claudino et al., 2015; Hepenstrick et al., 2012; Holderegger and Di Giulio, 2010; Hanski and Ovaskainen, 2003; Bailey et al., 2002). Several studies conclude that species richness and abundance are reduced within certain distances from linear infrastructure in general, and transportation infrastructure specifically, commonly referred to as the "road effect zone" (Benítez-López et al., 2010; Eigenbrod et al., 2009; Bissonette and Rosa, 2009; Biglin and Dupigny-Giroux, 2006; Pocock and Lawrence, 2005; Forman and Deblinger, 2000; Forman and Lauren, 1998; Reijnen et al., 1995). Substantial amounts of empirical research support the existence of effect zones (e.g., (Benítez-López et al., 2010; Bissonette and Rosa, 2009; Eigenbrod et al., 2009; Biglin and Dupigny-Giroux, 2006; Forman and Deblinger, 2000), and studies have estimated 15-20 % of the US (Forman and Lauren, 1998) and 16 % of the Netherlands (Reijnen et al. 1995) to be within effect zones. However, the causes for the observed effects on species richness and abundance are still under debate.

While species traits and attributes are likely to influence how they interact with transport infrastructure (Rytwinski and Fahrig, 2012a), possible causes for reduced species abundance and diversity frequently put forward in the literature are disturbances, e.g. traffic noise, vibrations and light pollution (Eigenbrod et al., 2009; Helldin and Seiler, 2003; Forman and Deblinger, 2000; Reijnen et al., 1996); road mortality (Summers et al., 2011; Fahrig and Rytwinski, 2009; Mumme et al., 2000); and habitat loss, fragmentation and barrier effects (Jaeger et al., 2007; Hanski and Ovaskainen, 2003; Hanski and Gilpin, 1991).

Road and railway effects manifest during different phases of the road/railway lifecycle. The construction and the localization and alignment of the corridor determine the levels of habitat loss and fragmentation, but also the creation of new habitat. New transport infrastructure environments can resemble biotopes such as natural grasslands and sandy and gravelly environments. Several rare plant species that would otherwise be found in habitat types of increasing rarity, have found refuge in road side verges and switchyards, along with high invertebrate diversity attracted to plant species, such as pollinators (Lennartsson and Gylje, 2009; Hovd and Skogen, 2005).

Being of linear shape, road and railway constructions consequently sub-divide habitat into smaller fragments. In addition to the conversion of previous habitat into "infrastructure corridor" habitat, the sub-division increases the proportion of edge relative to the core of the surrounding remnant habitat, which adversely impact on core habitat dwelling species (Marcantonio et al., 2013; Forman et al., 2003; Bender et al., 1998). The fragmentation effect of roads and railways is also considered to be further increased by its barrier effect, and the two combined are repeatedly suggested to be a main cause for declining wildlife populations (Claudino et al., 2015; van der Ree et al., 2009; Frair et al., 2008; Shepard et al., 2008; Jaeger et al., 2007).

The physical and technical design, size and width of the corridor define its level of interference with existing hydrological processes in the area. Steepness of the road or railway bank and surrounding ditches might reduce or accelerate hydrological flows, and at the landscape scale road networks interact with stream networks and might increase stream drainage density and overall peak flow (Kalantari et al., 2014; Coffin, 2007). Road and railway environments also contribute to erosion and sedimentation rates at the catchment scale, due to the abundance of exposed soil in the corridor and on road surfaces, from which several chemical contaminants and heavy metals also are transported by run-off into stream networks, lakes, wetlands and ponds (Forman et al., 2003; Sriyaraj and Shutes, 2001; Perdikaki and Mason, 1999). The physical design of the corridor also determines its barrier effect to a large extent for species with limited movement capacity such as amphibians, reptiles and small mammals, and together with fencing also for larger, wider ranging mammals. It is however likely that the barrier effect is possible to mitigate or compensate. For example, studies show that both under and over-passes, if well designed, are used by large as well as small animals even though their effectiveness for biodiversity conservation remains uncertain (Corlatti et al., 2009; Mata et al., 2009; van der Ree et al., 2009, 2007). Linear landscape elements (natural as well as human created) also function as dispersal conduits for some species, sometimes to the benefit of invasive species (Hulme, 2009; Stohlgren and Schnase, 2006; Yan et al., 2001). It has also been argued that the existence of transportation infrastructure indirectly drives biodiversity loss by increasing human access, resource extraction, and by providing incentives for different land uses; activities that in turn can have negative consequences for biodiversity (Freitas et al., 2010; Fu et al., 2010; Olsson, 2009; Balfors et al., 2005).

The effects on biodiversity of utilization and maintenance of roads and railways concern chemical contamination of air, water and biomass through vehicle exhaust, wear and tear, and use of pesticides and de-icing salts. Research in these areas suggests that impacts, e.g. bio-accumulation in plant species, remain within the road corridor in most cases (Munck et al., 2010; Bäckström et al., 2003; Forman et al., 2003; Jones et al., 2000; Trombulak and Frissell, 2000). The effects of traffic noise have been thoroughly studied, and are in several studies argued to be the cause for depressed densities and diversity of breeding birds and amphibians in areas close to roads (Francis et al., 2011; Hoskin and Goosem, 2010; Helldin and Seiler, 2003; Forman and Deblinger, 2000; Reijnen et al., 1996). Disturbance effects can nevertheless be mitigated by, for example, enforcing lower vehicle speed or installing noise screens in sensitive areas.

An evident effect of roads and railways is animal mortality, eloquently demonstrated by animal-vehicle collisions. Some researchers highlight road mortality as the primary cause for depressed densities as well as for loss of genetic variation in wildlife populations (Jackson and Fahrig, 2011; Summers et al., 2011). Road mortality is clearly a premier source of mortality for some species, but there is no consensus regarding the significance of the effects on the population level. It is likely that some species are more prone to get hit by vehicles than others (Rytwinski and Fahrig, 2012), and there is much support for the view that road mortality might function as population sinks for some species and cause local extinctions (McCall et al., 2010; van Langevelde et al., 2009; Row et al., 2007; Huijser and Bergers, 2000; Mumme et al., 2000; Fahrig et al., 1995).

Mitigation of road mortality is a rapidly developing research area of road ecology. Studies show that wildlife crossing structures, whether they be drainage culverts, ecoducts, over or under-passes, are used by the intended species, but their effectiveness in mitigating mortality (and also barrier effects) is unclear (Ascensão et al., 2013; Feurich et al., 2012; Diaz-Varela et al., 2011; Corlatti et al., 2009; Glista et al., 2009; Mata et al., 2009; van der Ree et al., 2009, 2007; Clevenger et al., 2001). Recently published studies on road and railway effects on wildlife populations demonstrate how the genetic indicators can be used to infer about population level effects. These methods show high potential for further reducing uncertainty on issues such as fragmentation, barrier effects and road mortality (Manel and Holderegger, 2013; Hepenstrick et al., 2012; Holderegger and Di Giulio, 2010; Corlatti et al., 2009).

4.2. The treatment of biodiversity in environmental assessment (Paper I)

4.2.1. Literature review on the treatment of biodiversity in EIA and SEA

The literature review on biodiversity and ecology in EIA and SEA revealed that the treatment of biodiversity impacts in EIA and SEA has been criticized from mainly three perspectives; 1)

EIS/ER contents, 2) ecological knowledge, and 3) methods (Table 3). The EIS/ER content problems concerned how well the EIS/ERs met the requirements of the EIA and SEA directives, i.e. if all the pieces of an environmental assessment as laid down by legislation could be identified in the EIS/ERs. The ecological knowledge problems concerned the level of understanding of transport infrastructure-ecology interaction expressed through the EIS/ERs, and the method related problems concerned the methods, data and tools used for biodiversity impact analysis. Of the 17 problem categories identified in the literature, six concerned EIS/ER contents, seven concerned ecological knowledge within the reports, and four concerned methods used for ecological impact prediction, assessment and evaluation. Problems frequently remarked upon were: low commitment to monitoring, assessment of impact significance, description of methods (related to the main problem category "contents"); indirect and cumulative impacts, consideration of habitat fragmentation, consideration of valuable non protected areas (related to "ecological knowledge"); the lack of use of quantitative methods, descriptive rather than analytical and predictive assessments (related to "methods").

4.2.2. Review of recent EIS/ERs

Three of the 17 problems identified in the literature review were not identified in any of the 23 reviewed EIS/ERs, 11 problems were identified in less than half of the documents, and three were identified in over half of the documents. The distribution of problems was not equal between the main problem categories 1) EIS/ER contents, 2) ecological knowledge and 3) methods. Compared to the problems frequently remarked upon in the literature, the results of the EIS/ER review show that the content related problems (main problem category 1) were identified in less than half to none of the reviewed reports. Regarding ecological knowledge (main problem category 2), the "treatment of habitat fragmentation" was found to be a problem in 43 %, "indirect and cumulative effects" in 48 %, and "consideration of valuable non-protected areas" in 17 % of the reviewed EIS/ERs. In addition, problems concerning "delimitation of the impact assessment study area" were found in 68 %, and "provisioning of necessary ecological data" in 52 % of the EIS/ERs. Regarding problems related to the methods used for impact assessment (main problem category 3), the problem category "assessments being descriptive rather than analytical and predictive" was found in 48 %, and "the scarce use of quantitative methods" was found in 87 % of the reviewed EIS/ERs (Table 3).

4.3. The Swedish road effect zones (Paper II)

The results show that natural grasslands and southern broadleaved forest are proportionally more affected by roads than coniferous forest and trivial broadleaved forest in Sweden, and that mammals in general are more negatively affected than birds. Figure 10 illustrates the predicted effects on mammals and birds within these habitat types. These figures show the proportion of the five habitat

Table 3. Summary of the results from Paper I. The literature review column shows which problem categories (originally 17 problems) were remarked upon in at least four articles and/or in articles published with at least 10 years in between. The EIS/ER review column shows which of the problems were identified in at least half of the reviewed EIS/ERs.

Problem		Literature review	EIS/ER review
EAR contents	Commitment to monitoring	X	
	Assessing impact significance	X	
	Description of methods	X	
Ecological knowledge	Consideration of habitat fragmentation	X	
	Indirect and cumulative impacts	X	
	Delimitation of the impact assessment study area		X
	Consideration of valuable non-protected areas	X	
	Provisioning of necessary ecological data		X
Methods	Use of quantitative methods in assessment of ecological impacts	X	X
	Descriptive rather than analytical and predictive assessments	X	

types within the four effect intensity zones, where the MSA of birds and mammals in an undisturbed area was close to 1 while a low MSA was interpreted as high road effect intensity.

Both graphs show that a larger proportion of natural grasslands (27.4 % for mammals and 4 % for birds) and southern broad leaved forest (33.2 % for mammals and 5.4 % for birds) were situated in zones with a predicted MSA < 0.5, i.e. with the highest road disturbance intensity, compared to trivial broadleaved forest (6.3 % for mammals and 0.8 % for birds), pine forest (7.5 % for mammals and 0.9 % for birds) and spruce forest (6.5 % for mammals and 0.6 % for birds). The results further show that almost twice the area of natural grasslands (27 - 50 %) and southern broadleaved forest (30 – 60 %) would be qualitatively altered by road effects so that they would support 50 – 65 % fewer mammals, compared to areas that were the least, or not at all, qualitatively altered (effect intensity zone of 0.9 - 1.0, 13.3 % of natural grasslands; 9 – 22 % of southern broadleaved forest).

For birds, these results suggest that the habitat types of natural grassland and southern broadleaved forest are proportionally more affected by roads than the other three habitat types (area within

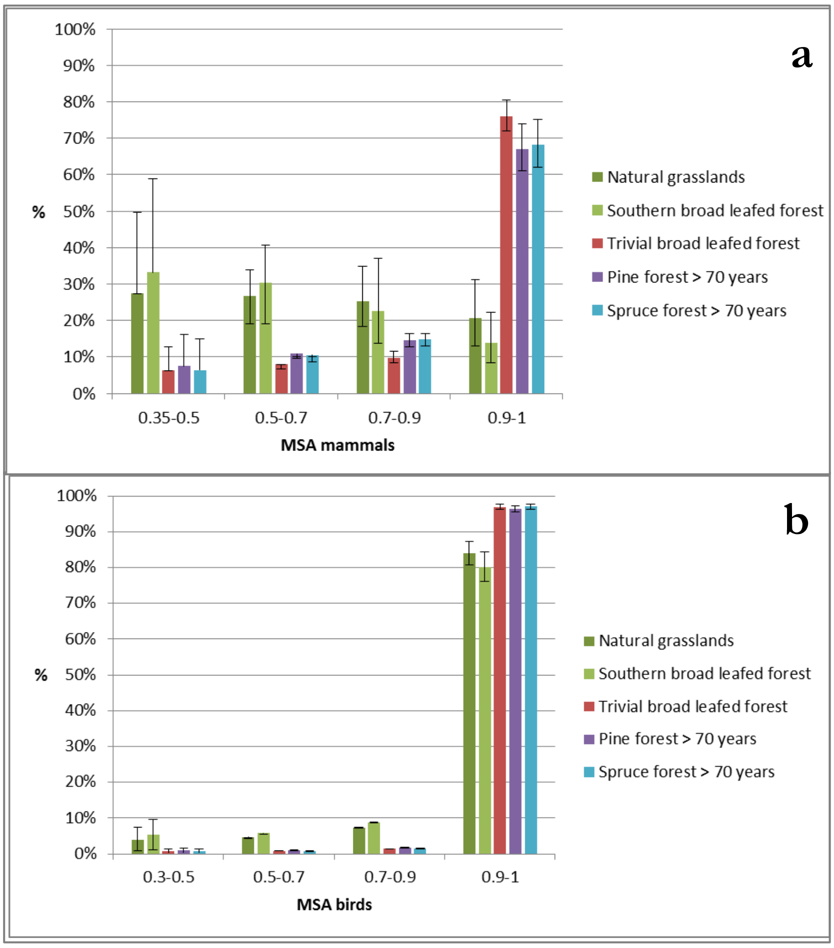

Figure 10. Results from the national scale analysis in Paper II. Graph a) shows the proportion of five valuable habitat types within the different road effect intensity intervals, as predicted for mammals, and graph b) shows the effect as predicted for birds.

MSA < 0.5; natural grassland 0.5 – 7 %, southern broadleaved forest 1 – 9 %, trivial broadleaved forest < 2 %, pine forest < 2 %, spruce forest < 2 %), but the overall impact on birds was less than on mammals.

The results from comparing fragmentation and disturbance on six ecological profiles showed habitat loss to be a generic impact, either through removal or through qualitative alteration. Disturbance generated a stronger change of the metrics than fragmentation in all profiles. Pure fragmentation effects, i.e. a loss of area together with an increase in number of patches in the habitat network, were minor and only detected in three profiles (high and low demand grassland profile; low demand forest profile, Fig. 11). The results further suggested that grassland birds were slightly more negatively affected than forest birds, and forest mammals with intermediate to high area demands were the most affected among the analysed ecological profiles. The CA and NP of this ecological profile were reduced by more than 0.5, relative to a road-less scenario (Fig. 11).

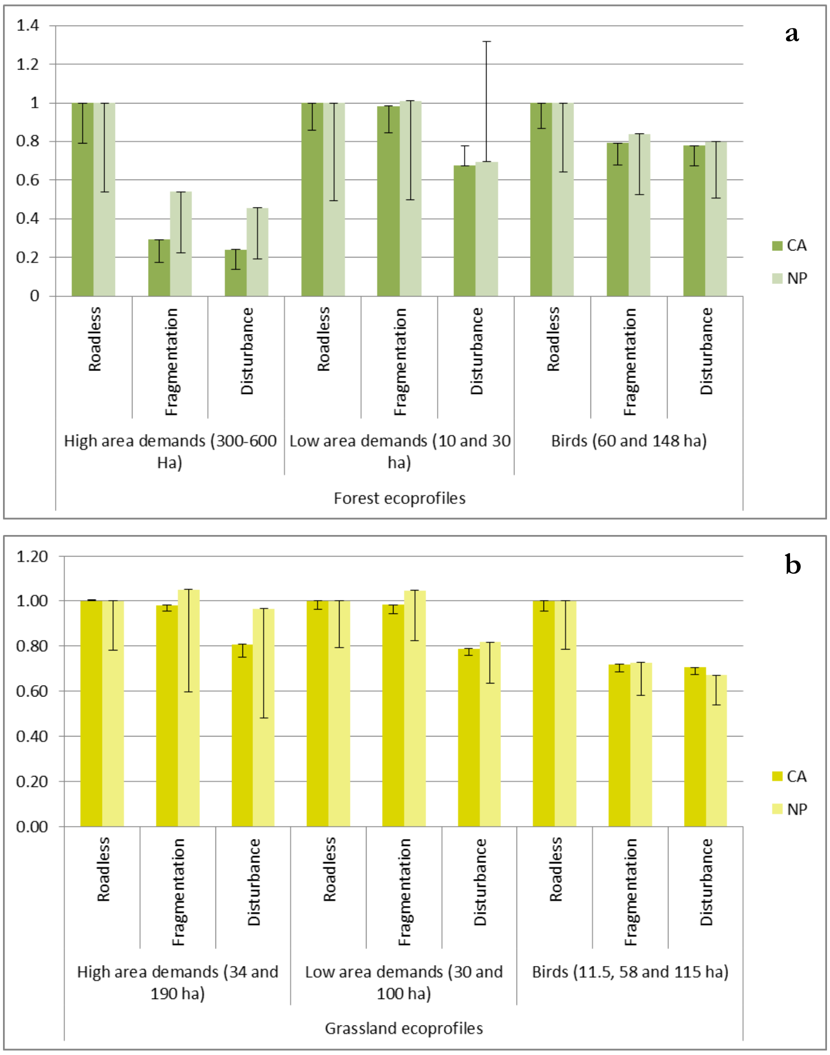

Figure 11. Results from the comparative analysis of fragmentation and disturbance in Paper II. Result are displayed as deviations from the road less habitat network created using the smaller radii. Graph a) shows the effect, as changes in total habitat area (CA) and number of patches (NP), on the forest ecological profiles, and graph b) shows the effect on the grassland ecological profiles.

4.4. Planning and evaluation of railway corridors (Paper III)

The railway corridor design model resulted in six corridors labelled *Eco_1*, *Eco_2*, *Equal_1*, *Equal_2*, *Geo_1*, and *Geo_2*. The variation in performance of the corridors was quite small. The corridor length varied from 13.0 - 14.2 km, the mass-balance calculations showed a deviation of the bank height from the mean m.a.s.l in the study area ranging from 2.3 - 3.8 m, the habitat loss varied between 0.06 - 1.88 ha, the relative connectivity loss caused by blocking dispersal pathways varied between 0.0 - 0.08 %, and the connectivity loss measured as "Accumulated connectivity loss"

Table 4. Overall scores for the six alternative corridors, on a 0-1 scale. The top-performing corridors are coloured green.

Corridor		Eco_1	Eco_2	Equal_1	Equal_2	Geo_1	Geo_2
Perspective	Ecological	0.559	0.619	0.380	0.426	0.175	0.343
	Geological	0.369	0.470	0.251	0.754	0.080	0.763
	Neutral	0.495	0.606	0.329	0.590	0.125	0.549

varied between 4727 - 6403 cost units. The variations were even smaller when comparing the performance of the corridors beginning at Origin 1 and Origin 2, respectively. The corridors beginning at Origin 1 were longer and had a poorer mass-balance performance. It was also clear that none of the corridors beginning at Origin 1 intersected a predicted movement pathway, and that the connectivity loss caused by the three corridors beginning at Origin 1 was negligible (< 0.1 %).

The overall scores for the six alternative corridors are presented in Table 4 and the top-rated corridors are illustrated in Figure 12. With the "ecological perspective" set of weights, the corridor with the highest overall score was *Eco_2*. With the "geological perspective" set of weights, the preferred corridor was *Geo_2*. If the "neutral perspective" set of weights would represent a compromise between the previous two, the preferred corridor would be *Eco_2*.

The "ecological perspective" preference *Eco_2* ranked highest from both the ecological and the neutral perspectives, but ranked as third choice from the geological perspective. *Geo_2* was the first choice from the geological perspective, while it was the fourth choice from the ecological perspective and the third choice from the neutral perspective. The sensitivity analysis revealed that the *Eco_2* corridor was a stable choice from an ecological perspective, but still sensitive to changes in the performance calculations and weights assigned to the criteria "Habitat loss" and "Broken links". Being the first choice from the "neutral" perspective as well, the sensitivity analysis revealed that *Eco_2* had an overall higher sensitivity to changes in performance and weights compared to the ecological perspective, specifically in the criteria "Corridor length" and "Habitat loss".

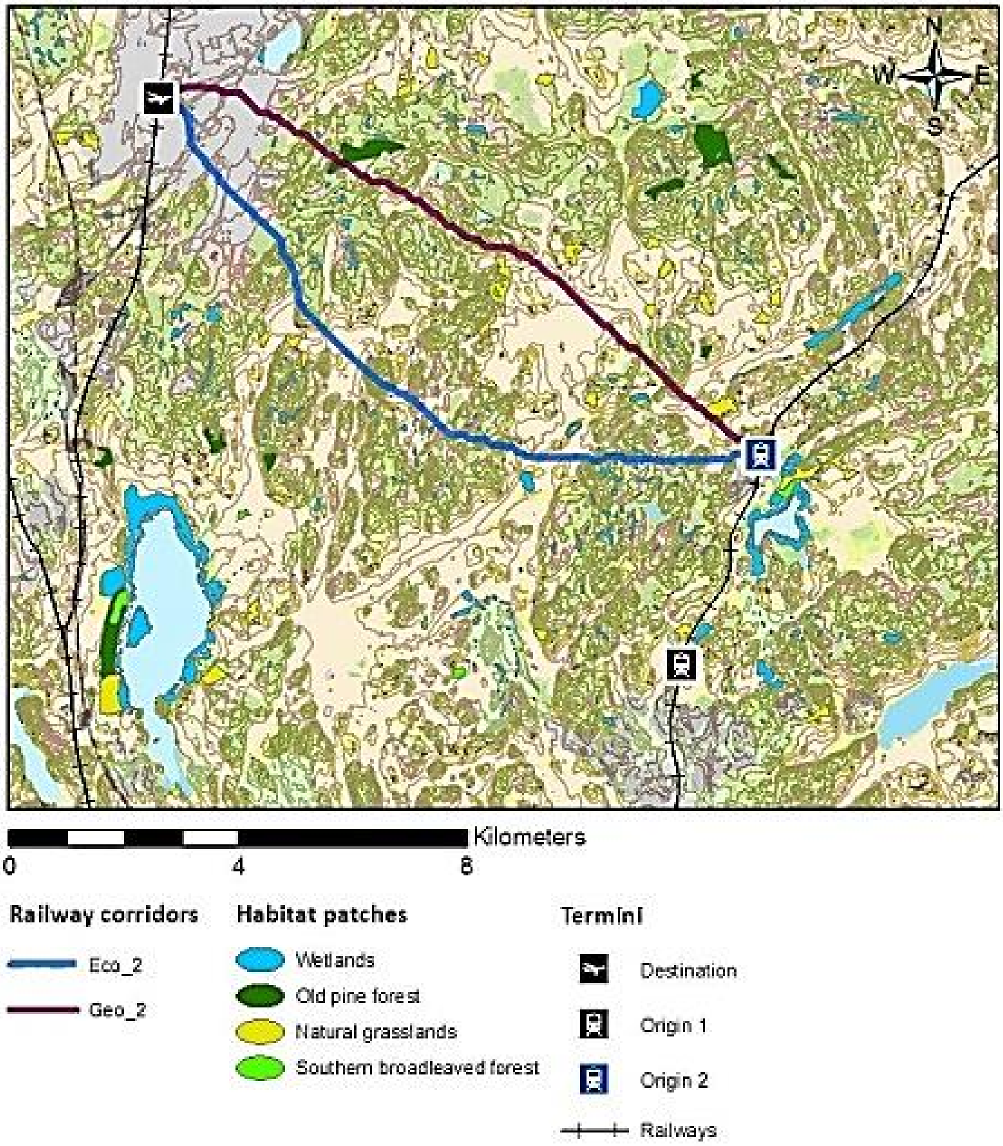

Figure 12. The results of the MCA-evaluation. The corridor with the highest overall score according to the three perspectives "ecology" (Eco_2), "geology" (Geo_2) and "neutral" (Eco_2). Elevation is represented by 4 m contours. Spatial habitat data previously published in (Mörtberg et al., 2012, 2007). Spatial data ©Lantmäteriet i2012/920.

4.5. The effects of barriers, landscape composition and fauna passages on habitat connectivity (Paper IV)

We observed that connectivity increased log-linearly with number of fauna passages, approaching a plane between five to seven passages (Fig. 13 a). Aggregation appeared to have no consistent effect on connectivity, and contrast appeared to greatly increase the variation in the results (Fig. 13 b and c).

Hypothesis 1: Several small passages increase connectivity more than a single large

We observed a strong positive relationship between overall connectivity, measured as the sum of mean current, and number of fauna passages across both the low contrast (M_{AGR}) and the high contrast data sets (M_{CONT}) (Fig. 13 a). The Kruskal-Wallis test returned highly significant for both the M_{AGR} dataset (Kruskal-Wallis $H = 39.11$, $df = 6$, $p < 0.001$) and the M_{CONT} data set (Kruskal-Wallis $H = 28.92$, $df = 6$, $p < 0.001$), suggesting that the observed increase in connectivity would be due to the addition of fauna passages in the barrier. The Mann-Whitney U-test between the different SLOSS-scenarios (post-hoc test) showed a significant current increase at five to seven small passages relative to a single large ($p < 0.05$). The same test between the contrast-enhanced SLOSS scenarios returned significant results at four to seven passages relative to a single large ($p < 0.05$). These results suggest a rejection of h_{null} and that the construction of several small fauna passages would provide better opportunities for genetic exchange across a barrier than a single large passage.

Hypothesis 2: Increasing contrast might increase or decrease connectivity

The two tailed Mann-Whitney U-test between the samples M_{AGR} and M_{CONT} was not significant (Mann-Whitney $U = 668.5$, $n1 = n2 = 42$, $p = 0.056$), but the same test between the SD-datasets (SD_{AGR}; SDC_{ONT}) was significant (Mann-Whitney $U = 0$, $n1 = n2 = 42$, $p = < 0.001$). We can also observe from descriptive statistics that the variance in the M_{CONT} is greater than in M_{AGR} (Fig. 13 c).

Hypothesis 3: Aggregation has an effect on connectivity in contrast rich landscapes

The regression analysis results showed an insignificant relationship between connectivity and aggregation level. The descriptive plots of connectivity as a function of aggregation level also indicated that there was no such relationship (Figure 13 b). From these results, we could not conclude that there was any effect of aggregation on connectivity.

5. DISCUSSION

5.1. The treatment of biodiversity in environmental assessment (Paper I)

Of the 17 problem categories concerning the treatment of biodiversity in EIA and SEA identified in the scientific literature, six concerned the EIS/ER contents. The result of the EIS/ER review showed that the content related problems were reported in less than half to none of the EIS/ERs. This is a strong indication of an improvement since Treweeks' (1996) article on ecology in EIA, where several content-related problems were listed. For instance, problems concerning the characterization of baseline conditions and description of mitigation measures could not be identified in any EIS/ER reviewed in the current study (Paper I). The results also showed a considerable improvement in the commitment to monitoring activities.

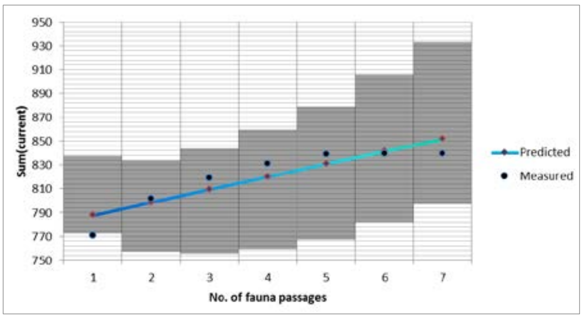

Figure 13 a. Number of fauna passages as explanatory variable for current increase, interpreted as connectivity increase. The shaded area represents a 95 % confidence interval.

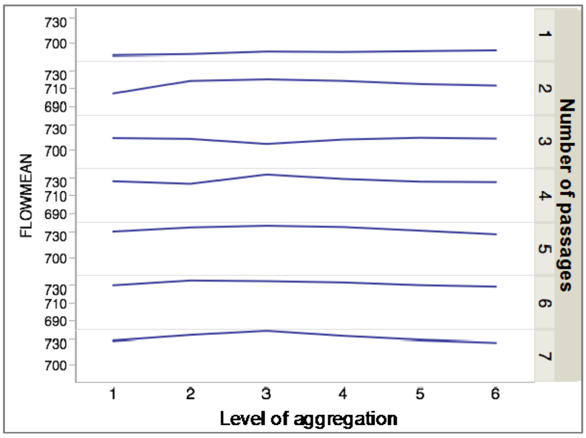

Figure 13 b. Mean connectivity in the test landscapes explained by level of aggregation. The effect of increasing aggregation remained negligible.

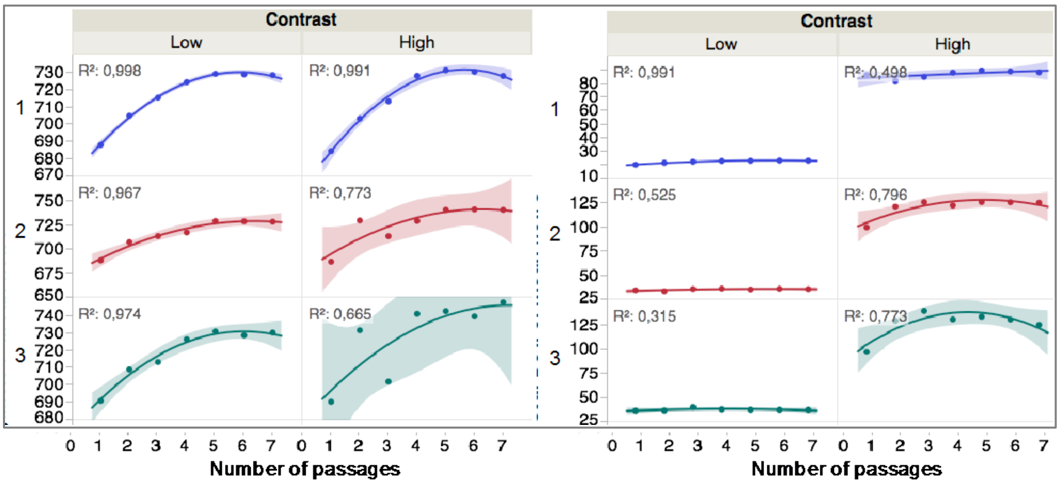

Figure 13 c. The effect of contrast on the variance in the results. To the left we see the difference in mean connectivity measurements between the low (M_{AGR}) and high contrast (M_{CONT}) data sets, and to the right we see the difference in standard deviation.

Of the problems related to ecological knowledge (main problem category 2), the EIS/ER review showed that the treatment of habitat fragmentation was considered to be a problem in slightly less than half of the reviewed EIS/ERs. There was an awareness of the negative effects of fragmentation in most EIS/ERs, but the effects were described in general terms, vaguely related to the project and only qualitatively analysed in a few, and not quantified in any. Since habitat fragmentation and changes in connectivity can be seen as major impacts of transport infrastructure (Benítez-López et al., 2010; Forman et al., 2003; Jaeger et al., 2007), and indirect and cumulative impacts can be anticipated this can be seen as a remaining critical issue.

Problems with the delimitation of the impact assessment study area were identified in more than half of the reviewed EIS/ERs (Table 3). The potential to successfully address many indirect impacts from transport infrastructure is highly dependent on the delimitation of the study area and the time frame specified for the impact assessment study. Thus, if this would be revised properly, it could generate improvements in the treatment of ecological impacts in EIA and SEA. The ERs (related to SEA) often covered a geographical area large enough to encompass important ecological processes as well as for indirect impacts to be included in environmental assessment. The temporal scale in these ERs was set by the end date of the plan, which was commonly specified to span one to three decades. By contrast, in the reviewed EISs (related to EIA), the study areas were defined as the infrastructure corridor, and a time frame for the impact assessment was rarely specified. Even though it is difficult to determine the delimitation of the areas that are required to assess ecological impacts, it can be concluded that SEA, through a more comprehensive approach, provides better opportunities for predicting impacts on ecological processes, than EIA with relatively narrow approaches and study areas.

Regarding the methods used for impact assessment (main problem category 3), results showed that the problems of assessments being descriptive rather than analytical and predictive as well as the related use of quantitative methods remained problematic. Ecological impacts were qualitatively predicted in all but one EIS/ER and assessments and evaluations were communicated in written text and by means of qualitative matrices. The analytical quality of these descriptions varied greatly. In some reports impacts were merely listed without being linked to the project context, whereas other reports described likely cause-effect relationships as well as uncertainties and scale issues. Impacts were identified, predicted, and evaluated by the use of similar types of qualitative matrices and criteria. Therefore, the variation in the quality of assessment likely depended on the input data itself (e.g. expert opinions, literature studies, field studies) or on the interpretation of the input data, and not on the methods used per se.

Quantitative predictions were only presented in one of the reviewed EIS/ERs, to estimate the effects on aquatic flora and fauna from increased loads of de-icing salts (produced in Sweden 2012). The scarcity of such quantitative methods was considered to be a problem already by Treweek (1996) as well as by Geneletti (2006) and Gontier et al. (2006), and can thus still be considered to be problematic. Similar critique was given by Geneletti (2006), who concluded that the use of GIS was limited to mapping and display.

The spatial distribution of impacts together with the delayed and non-linear response to change by many recipients, species or ecosystems (Peters et al., 2004; Findlay and Bourdages, 2000; Trombulak and Frissell, 2000; Holling, 1973), can be seen as a significant challenge to the prediction of ecological impacts, not at least for EIA and SEA. In order to improve the treatment of ecological impacts in EIA and SEA, a way forward may be to standardize the spatial and temporal scales used for impact prediction and assessment, follow-up studies and monitoring. An increased use of quantitative methods together with robust frameworks for quantitative ecological analysis and interpretation, including thresholds and reference values, would facilitate and mainstream ecological assessment.

5.2. The Swedish road effect zones (Paper II)

The results of the national scale road effect modelling (Study Area A) showed that natural grasslands and southern broadleaved forest were the habitat types of high biodiversity value that were most exposed to road effects in Sweden (Fig. 9). This makes them suitable targets for mitigation of road effects and less suitable habitat types for new road corridors. The studies underpinning the MSA predictions were carried out in environments and on species groups similar to Swedish conditions, flora and fauna (Benítez-López et al., 2010). However, the predicted road effects were very coarse, and some of the assumptions can be questioned. For instance, other forms of impacts (e.g. from forestry or mining) were not taken into account, and in addition to the modelled road network, there is an extensive network of forest, municipal and private roads that were not considered in this analysis.

The adaptive capacity of mammals and birds should not be underestimated, and it is likely that areas exposed to moderate and low disturbance could still constitute suitable habitat for many species. However, the results still describe a situation where around 80 % of natural grasslands and southern broadleaved forest, and between 20 % and 30 % of the other three habitat types of high biodiversity value, would constitute slightly to highly affected mammal habitat. Together with the substantial volumes of empirical research that found differences in the environmental characteristics between areas remote from and areas in the vicinity of transport infrastructure (e.g. (Benítez-López et al., 2010; Bissonette and Rosa, 2009; Coffin, 2007; Biglin and Dupigny-Giroux, 2006) and it may not be unlikely that the magnitude and spatial distribution of the actual road effects would be within the CI of the predicted MSA. The results of this study therefore be viewed as an indication of the spatial extent of the Swedish road effect zone.

The study of fragmentation and disturbance showed that habitat loss was a pronounced result of both effects, and that pure fragmentation effects were minor in comparison with disturbance effects. A reduction in CA (habitat loss) could occur either from physical removal or through a degradation of suitability. Habitat loss was thus an expected impact from both fragmentation and disturbance. Fragmentation, as defined in Paper II, could only occur as a sub-division of patches; as an increase in NP together with a decrease in CA (the study design would not allow habitat to increase). In all profiles, disturbance had a stronger effect, expressed as reduced CA, on the habitat networks than fragmentation (Fig. 10), suggesting that habitat quality would be an important factor for overall habitat availability in a landscape. Fragmentation effects were present in 3 profiles (Fig. 11, low demand forest and high and low demand grassland profiles), but the effects were minor contrary to expectations. The low demand forest profile however, expressed in response to disturbance an increase in NP with 0.32 points, relative to the road-less scenario. A potential cause for this result could be that the preferred habitat of the low demand forest profile was spatially aligned with the road network in a way that, when disturbance effects were introduced, multiple habitat patches were subdivided. It is likely that for species which perceive the landscape as a patch-network of habitats, there would be thresholds in network connectivity beyond which distinguished changes, i.e. as expressed by the low demand forest profile, could occur (Frair et al., 2008).

Road effects had strong impacts on the forest ecological profiles, especially on the mammal profile with high area demands (Fig. 10). This is in line with other studies suggesting that species with high area demands, like large ungulates and carnivores, are especially sensitive to transport infrastructure effects like barriers to movement, disturbance and mortality (Rytwinski and Fahrig, 2012b; Jaeger et al., 2005). For the grassland ecological profiles, road effects also had substantial impacts, and the bird profile was specifically affected (Fig. 10).

For all ecological profiles, the road effects imposed habitat loss of homogenous habitat as well as loss of entire habitat patches, both of which would increase extinction risks of local populations. A Species movement capacity determines the potential for a species to use scattered habitat patches as well as to migrate between populations and to colonize empty suitable habitat, two crucial processes which arguably determine the viability of populations in fragmented landscapes (Holderegger and Di Giulio, 2010; Hanski and Ovaskainen, 2003; Hanski and Gilpin, 1991). Movement capacity was in this study indirectly approximated by the interval between the two assumptions on home range sizes and by calculating the Mean-statistic in creation of the habitat networks (See Paper II for details). The assumptions on home range size, habitat suitability classification as well as the cut-value (0.5 suitability on a 0 - 1.0 scale) used to spatially delimit patches constitute the major uncertainties of this study and would need further investigations.

Other issues implicit in habitat suitability modelling is the degree to which the habitat preferences of a model species can be described by available habitat data (Braunisch et al., 2008; Rubino and Hess, 2003) and aggregation rules for species movements (Laitila and Moilanen, 2013). These uncertainties call for increased model testing and evaluation, whether it be empirical models driven by occurrence and movement data of selected focal species or expert models built from scientific knowledge (Bradley et al., 2012; Gontier et al., 2010; Elith and Leathwick, 2009; Rubino and Hess, 2003).

Another issue is how to best demonstrate the integrity of ecological (habitat) networks and the response of an ecological profile (or model species) to landscape changes. CA and NP were used in this study as these metrics represent habitat amount and fragmentation, and relate to the Species-Area relationship which have been used in conservation biology to address consequences from e.g. habitat loss and in prioritization of conservation areas and actions (e.g. Smith, 2010). A multitude of landscape metrics have been suggested as predictors of the integrity of ecological networks (Walz and Syrbe, 2013; Zetterberg et al., 2010; Saura and Torné, 2009; Cushman et al., 2008; Schindler et al., 2008). The modelling carried out in this study illustrated how road effects can be quantified and assessed, and how meta-population theory and metrics can be mobilized to benefit environmental assessment of transportation networks. The results identified valuable nature types and groups of species likely to be adversely impacted by road effects. Such information is valuable for conservation planning as well as for the development of regional to national scale transportation strategies. The results may also be used to inform further analysis and modelling. The habitat networks created could, depending on the research question, be input to graph theoretical habitat models for estimation of i.e. patch level conservation values (e.g. Zetterberg et al., 2010) or for evaluation of alternative construction projects (Paper III).

5.3. Planning and evaluation of railway corridors (Paper III)

Ecological and geological criteria derived from spatial modelling were integrated into an MCA framework and six railway corridor alternatives were designed and evaluated. The methodology allowed for taking advantage of existing data, models and expert judgements which were transformed into suitability criteria for design and evaluation, and combined through a set of decision rules. A strength of the MCA framework is that once it is set up, it can readily be re-iterated using different weighting schemes, which allows for sensitivity analyses in order to evaluate the robustness of the applied decision rules and of the ranking of the alternative railway corridors.

Both the ecological and the geological criteria were based on available data, models and expert knowledge, and each criterion would benefit from an enhanced knowledge base on how to best support sustainable railway construction. A conflict between conserving habitat vs conserving movement pathways between habitats was identified in the case study application of the MCA framework, which highlights the complexity of meeting multiple objectives even within one group (the ecological) of sustainability criteria. This conflict would need to be addressed in a real planning case as well, as a corridor that does not run across habitat might still interfere with movement and dispersal pathways between habitat areas.

For the design of railway corridors, it is possible that synergies between ecological and geological criteria may emerge. The case study application of the MCA framework revealed a positive synergy in the study area between the geological criteria "mass-balance performance" and the three ecological criteria ("Habitat loss", "Broken links" and "Accumulated connectivity loss"). Other positive synergies between disparate sustainability criteria could likely be detected using the MCA framework in other landscapes and planning proposals. For example, the construction of a railway tunnel may also help to maintain habitat connectivity, which could be compared to the alternative which would be a chasm or canyon that constitute a barrier to non-flying animals. Another example would be a bridge over a valley for the railway, maintaining habitat connectivity beneath it, compared to the alternative of filling up the valley with ballast material. In both examples, ecological and geological considerations may gain from the same solution, which would provide interesting opportunities to meet multiple sustainability objectives.

In a real transport infrastructure planning case, other sustainability issues than ecological and geological should be taken into account, such as recreation, cultural landscapes and other ecosystem services (see e.g. Mörtberg et al., 2007). How to weigh the design criteria in order to fit a particular decision situation is an issue that was not included in this study, but that would require much attention if the planning model would be applied to a real context. The MCA framework is however open for integrating such aspects into transport infrastructure planning, which makes planning and related environmental assessments suitable areas for application of the demonstrated methodology.

5.4. The effects of barriers, landscape composition and fauna passages on habitat connectivity (Paper IV)

The results suggested that the number of fauna passages and the level of contrast in the underlying landscape were important considerations for restoring connectivity across an impermeable barrier. Aggregation appears to be of low importance for connectivity unless there is a minimum level of contrast between habitat types. In such landscapes, aggregation likely matters for where a fauna passage should be located along a barrier, as the size and distribution of patches determine where high and low resistance areas would be located. A fauna passage located in a high resistance area may greatly reduce the effect of that passage, or may even increase the accumulated resistance relative to another scenario. Inversely, a fauna passage providing passage through a low resistance area greatly increases the total flow in the entire landscape, and would thus be more effective compared to the previous situation.

The realism of the chosen representation of connectivity (as a function of resistance) could be discussed. While the Circuitscape model have performed well compared with other common dispersal models (McRae and Beier, 2007), habitat connectivity models based on probabilistic or binary representations of connectivity have also displayed good fit with species occupancy data (Rodríguez - Pérez et al., 2014; Awade et al., 2012; Pereira et al., 2011; Ribeiro et al., 2011) and genetic datasets (Neel, 2008). Common ecological metrics for species movement and dispersal essentially rely on distances between populations or (potentially occupied) habitat patches, weighted by some factor of importance; e.g. maximum dispersal capacity or arrival probabilities (Saura and Pascual-Hortal, 2007; Pascual-Hortal and Saura, 2006) or expert models on habitat preferences (Gontier et al. 2006) (Paper II). The circuit theory model circumvents the "single path" problem of these metrics (Pinto and Keitt, 2009; McRae, 2006), and allows for multiple dispersal pathways to contribute to connectivity. This improves the ecological realism of circuit theory modelling compared to other approaches, but the connectivity metrics deduced from circuit theory still rely on a classification of the landscape into dispersal resistance and will consequently reproduce the uncertainties and potential errors related to the designation of resistance values.

The results of this study remain general, but indicate that several small fauna passages would increase connectivity more than a single large for species to who the cost of traversing the given landscape and the barrier matters. The post hoc analysis returned a significant connectivity increase for at least four small passages relative to a single large passage but as the descriptive graphs (Fig. 13 a) illustrate, connectivity would increase substantially already at two passages relative to a single large. However, these results should be viewed in light of the simplified context provided by the model landscapes. The topology, land use, natural barriers and distribution of habitat types might constrain the possible number of crossing structures to less than four, or require even more of them to achieve increased flow across a barrier. These results also indicate that the effectiveness of a fauna passage most likely will depend on its location along the barrier. The effect of contrast highlights that effective barrier mitigation would have to be context adapted, and that a standardized solution, e.g. even spacing between fauna passages across a motorway, might not yield the expected results in landscapes perceived as contrast rich by the target species for mitigation. Constructing a fauna passages in areas perceived as poorly connected or even unconnected might be highly ineffective relative to locating the same construction in an area were the species would normally try to cross the barrier. Thus, for barrier mitigation planning to be effective, it will be necessary to identify a set of relevant mitigation target species, i.e. ecological profiles (Karlson and Mörtberg, 2015; Mörtberg et al., 2012; Angelstam et al., 2004), for which the landscape can be represented in terms of aggregation, contrast, and further in terms of movement resistance.

5.5. Prospects for road ecology and environmental assessment

Road ecology will continue to rely to a great extent on basic research carried out specifically on the effects of transport infrastructure, but also on animal behaviour, meta-population theory, landscape ecology and on methodological developments in genetics, GIS-technology and spatial ecological modelling. The last decade has seen a rapid development of tools and methods that could be suitable for environmental assessment of road and railway plans and projects, not at least with regards to habitat loss and connectivity, and spatial decision support to physical planning (among others; Bottero et al., 2013; Etherington, 2012; Coutinho-Rodrigues et al., 2011; Gontier et al., 2010; Zetterberg et al., 2010; Saura and Torné, 2009; Shah and McRae, 2008; Jaeger et al., 2005). Still, field studies will continue to be a primary source of information concerning most issues, such as road mortality, behavioural responses of species, barrier effects on genetic exchange and others (Rytwinski et al., 2015). Field work and gathering of empirical data will also be crucial for validation and calibration of relevant models, as erroneous or highly uncertain model output would contribute little in terms of decision support concerning environment-development conflicts.

A future challenge however, and the suggested path forward, is for road ecology to become more deeply integrated into road and railway planning. According to e.g. Morrison-Saunders and Retief (2012), several important environmental concerns are often addressed before the EIA or SEA process has been initiated, which can be regarded as a main barrier for EIA and SEA to effectively deliver on their sustainability mandates. The influence of environmental assessment findings on the actual planning process as well as in decision making have also been considered to be limited (Jay et al., 2007). Effects with potentially detrimental impacts, such as fragmentation and barriers as pointed out in Paper I, are often difficult to address during an EIA process as ecologically important decisions, i.e. on corridor location(s) may already have been taken. Accumulative and network level effects, such as those that were quantified in Paper II, require large study areas and a sufficient time horizon, whereas an EIA is predominantly carried out in the near vicinity of the road or railway corridor with no specified time frame (Paper I; Geneletti, 2006). It is possible that for EIA and SEA to be able to address the aforementioned issues and fully realize their potential as instruments for sustainable development, EIA and SEA will have to evolve beyond impact assessment and come to encompass a larger proportion of the planning process (Azcárate, 2015).

The integration of road ecology into transportation planning is already a work in progress; see e.g. (van der Ree et al., 2015; Forman et al., 2003). However, the currently most emphasized issues in need of research require the integration of road ecological knowledge from the earliest possible stage of the planning process, to the implementation of post monitoring schemes (see e.g. Rytwinski et al., 2015). In other words, road ecological knowledge and best practices need to be institutionalized within road and railway planning as an intrinsic part of the planning process. In particular with regards to research on the effectiveness of (often expensive) road effect mitigation activities such as fauna passages, fencing and noise mitigation. The minimum requirements of statistical methods are that field sampling can be carried out before as well as after the installation of a mitigation measure, and preferably at several control locations. For all spatial ecological models (Papers II, III and IV), it is important to further increase the knowledge base on the needs of species and populations for persistence in the landscape, on the impacts of transport infrastructure as well as on the efficiency of different mitigation measures. It is therefore also important that relevant data, whether collected by authorities or researchers, is made available for future road ecology research.

An integration of road ecological knowledge into road and railway planning would strengthen ecological concerns which have not been successfully addressed in concurrent environmental assessment frameworks. It further presents itself as an opportunity to systematically identify cross-disciplinary benefits between road and railway sustainability concerns (Paper III). For instance, wetlands are often unsuitable areas for construction both from a geotechnical and an ecological perspective. The construction of bridges and tunnels may provide a faster and more direct transportation link between destinations and less of an impact on habitat connectivity compared to filling up valleys and severing hills. Drainage structures might also function as underpasses for amphibians and reduce the barrier effects for migrating fish (Feurich et al., 2012; Clevenger et al., 2001; Yanes et al., 1995) and road and railway corridors can be designed to support biodiversity and certain ecosystem services (Ottosson et al., 2012). It is likely that identifying such synergies at an early planning phase would result in improved environmental performance as well as reduced construction costs.

How EIA and SEA might evolve to address the problems pointed out in i.e. Paper I remains uncertain, but EIA and SEA will continue to be an opportunity for biodiversity impact assessment at the corridor or landscape scale where several positive as well as negative changes might occur. EIA and SEA forums also constitute platforms for innovation and information exchange between researchers and practitioners, most notably concerning methods and tools for biodiversity impact analysis. As demonstrated by this thesis, decision support on biodiversity issues can be achieved through a shift towards an increased use of quantitative assessment tools. The impacts of habitat loss and changes to habitat connectivity, which have been difficult to account for in EIA and SEA so far, can be assessed using quantitative methods and used to inform design and evaluation of planning alternatives. At several stages in the road or railway planning process there is a need to assess impacts on biodiversity, thus environmental assessment practitioners should endorse a future use of ecological indices, metrics and tools for prediction and sensitivity analysis.

6. CONCLUSIONS

The overall aim of this thesis was to contribute to knowledge on the current state of applied road ecology, and with methodologies suitable for environmental assessment of road and railway impacts on biodiversity. A literature review revealed that environmental assessment of roads and railways is a complex task, as several important impacts occur indirectly, accumulate over time and manifest in areas far away from the road or railway. There is much consensus in the research community that habitat loss, fragmentation and barrier effects can have detrimental impacts on biodiversity, while there are also certain positive impacts. The results of the comparative analysis between fragmentation and disturbance (Paper II) indicated that qualitative degradation of habitat also might be a major impact of transportation networks.

The nature of ecosystem response to change, often non-linear and sometimes irreversible, is challenging for road and railway planning and environmental assessment. The review of environmental assessment reports produced in Sweden and the UK reflected this fact, but it also clear that biodiversity impact analysis in environmental assessment has improved continuously in this respect over the years (2006-2013). However, some problems with the treatment of biodiversity in EIA and SEA remain, i.e. that the environmental assessment did not encompass the relevant scales (both in time and space) across which ecological processes take place, and fragmentation effects and changes in habitat amount and connectivity were not properly analysed.

The research presented in this thesis demonstrated that there are tools available for biodiversity impact analysis which could improve the treatment of e.g. the above listed problems. Paper II demonstrated how road network effects can be calculated with available datasets, expert knowledge and existing biodiversity indicators and models. The analysis framework presented in Paper II could, for instance, benefit environmental baseline descriptions and the assessment of impacts of transportation plans or strategies.

Paper III further elaborated how scientific indicators and predictors for biodiversity can be mobilized, and how established analysis techniques can be combined to generate transparent and systematic decision support to road and railway planning. Paper IV demonstrated how a population genetic model can benefit barrier effect mitigation strategies by reducing uncertainty on important questions such as for which species, how many and where should fauna passages be built.

Spatial ecological models appropriate for planning and environmental assessment are developing rapidly and the EU Directives on EIA and SEA currently in effect encourage an increased use of such tools. Environmental assessment of road and railway impacts on biodiversity could be improved through an increased use of quantitative analysis tools, i.e. such as the ones demonstrated in this thesis. An increased use of quantitative tools will, however, require a future commitment to increase the knowledge on the habitat preferences, behaviour and factors influencing population persistence of suitable models species, and for this data to be available for further research and method development. There is also a need to consider some impacts on biodiversity before an EIA or a SEA process becomes a legal imperative. Therefore, to improve environmental assessment of road and railway impacts on biodiversity and facilitate future road ecology research, road ecological knowledge and best practices need further integration into road and railway planning.

7. Future research

The "road effect zone" concept has much potential in a planning and environmental assessment context. A revival of effect zone research with a focus on tools for application in environmental assessment and planning could be an important contribution to applied biodiversity impact analysis. Ecosystem services is a concept of increasing attention among EIA and SEA practitioners and planners, whereas road ecology research in an ecosystem services context is relatively scarce. There is also a general (but strong) need of verification and calibration of relevant spatial ecological models and methods, specifically of dispersal and movement models, species distribution models and habitat suitability models. The use of genetic markers and indicators also show much potential for future research to address a range of important issues and to improve the knowledge of e.g. barrier effects, fauna passage effectiveness and drivers for loss of genetic variation.

8. References

Angelstam, P., Edman, T., Dönz-Breuss, M., Wallis deVries, M., 2004. Land management data and terrestrial vertebrates as indicators of forest biodiversity at the landscape scale. Ecol. Bull. 51, 333–349.

Ascensão, F., Clevenger, A., Santos-Reis, M., Urbano, P., Jackson, N., 2013. Wildlife–vehicle collision mitigation: Is partial fencing the answer? An agent-based model approach. Ecol. Model. 257, 36–43. doi:10.1016/j.ecolmodel.2013.02.026

Awade, M., Boscolo, D., Metzger, J.P., 2012. Using binary and probabilistic habitat availability indices derived from graph theory to model bird occurrence in fragmented forests. Landsc. Ecol. 27, 185–198.

Azcárate, J., 2015. Beyond Impacts: Contextualizing Strategic Environmental Assessment to Foster the Inclusion of Multiple Values in Strategic Planning. TRITA-LWR PHD-2015:02 (Doctoral Thesis). KTH Royal Instititue of Technology, Stockholm, Sweden.

Bäckström, M., Nilsson, U., Håkansson, M., Allard, B., Karlsson, S., 2003. Speciation of heavy metals in road runoff and roadside total deposition. Water. Air. Soil Pollut. 147, 343–366.

Bagli, S., Geneletti, D., Orsi, F., 2011. Routeing of power lines through least-cost path analysis and multicriteria evaluation to minimise environmental impacts. Environ. Impact Assess. Rev. 31, 234–239. doi:10.1016/j.eiar.2010.10.003

Bailey, S.-A., Haines-Young, R.., Watkins, C., 2002. Species presence in fragmented landscapes: modelling of species requirements at the national level. Biol. Conserv. 108, 307–316. doi:10.1016/S0006-3207(02)00119-2

Balfors, B., Mörtberg, U., Gontier, M., Brokking, P., 2005. Impacts on region-wide urban development on biodiversity in strategic environmental assessment. J. Environ. Assess. Policy Manag. 7, 229–246.

Bastian, O., Steinhardt, U., Naveh, Z., 2002. Development and Perspectives of Landscape Ecology. Kluwer Academic Publishers.

Beier, P., Majka, D.R., Spencer, W.D., 2008. Forks in the Road: Choices in Procedures for Designing Wildland Linkages. Conserv. Biol. 22, 836–851. doi:10.1111/j.1523-1739.2008.00942.x

Bender, D.J., Contreras, T.A., Fahrig, L., 1998. Habitat loss and population decline: a meta analysis of the patch size effect. Ecology 79, 517–533.

Benítez-López, A., Alkemade, R., Verweij, P.A., 2010. The impacts of roads and other infrastructure on mammal and bird populations: A meta-analysis. Biol. Conserv. 143, 1307–1316.

Biglin, K., Dupigny-Giroux, L.-A., 2006. Mapping the road effect-zone to assess impacts of proposed road segments. J. Conserv. Plan. 2, 1–16.

Bissonette, J.A., Rosa, S.A., 2009. Road Zone Effects in Small-Mammal Communities. Ecol. Soc. 14, 1–15.

Bodin, Ö., Saura, S., 2010. Ranking individual patches as connectivity providers: integrating network analysis and patch removal experiments. Ecol. Model. 221, 2393–2405.

Bottero, M., Comino, E., Duriavig, M., Ferretti, V., Pomarico, S., 2013. The application of a Multicriteria Spatial Decision Support System (MCSDSS) for the assessment of biodiversity conservation in the Province of Varese (Italy). Land Use Policy 30, 730–738.

Bradley, B.A., Olsson, A.D., Wang, O., Dickson, B.G., Pelech, L., Sesnie, S.E., Zachmann, L.J., 2012. Species detection vs. habitat suitability: Are we biasing habitat suitability models with remotely sensed data? Ecol. Model. 244, 57–64.

Braunisch, V., Bollman, K., Graf, R.F., Hirzel, A.H., 2008. Living on the edge - Modelling habitat suitability for species at the edge of their fundamental niche. Ecol. Model. 214, 153–167.

Byron, H.J., Treweek, J.R., Sheate, W.R., Thompson, S., 2000. Road development in the uk: an analysis of ecological assessment in environmental impact statements produced between 1993 and 1997. J. Environ. Plan. Manag. 43, 71–79.

Catlow, J., Thirlwall, C.G., 1976. Environmental impact analysis. London.

Claudino, E.S., Gomes, M.A.F., Campos, P.R.A., 2015. Extinction debt and the role of static and dynamical fragmentation on biodiversity. Ecol. Complex. 21, 150–155. doi:10.1016/j.ecocom.2014.11.011

Clevenger, A.P., Chruszcz, B., Gunson, K., 2001. Drainage culverts as habitat linkages and factors affecting passage by mammals. J. Appl. Ecol. 38, 1340–1349.

Coffin, A.W., 2007. From roadkill to road ecology: A review of the ecological effects of roads. J. Transp. Geogr. 15, 396–406.

Corlatti, L., Hackländer, K., Frey-Roos, F., 2009. Ability of Wildlife Overpasses to Provide Connectivity and Prevent Genetic Isolation. Conserv. Biol. 23, 548–556.

Coutinho-Rodrigues, J., Simão, A., Antunes, C.H., 2011. A GIS-based multicriteria spatial decision support system for planning urban infrastructures. Decis. Support Syst. 51, 720–726.

Cushman, S.A., McGarigal, K., Neel, M.C., 2008. Parsimony in landscape metrics: Strength, universality, and consistency. Ecol. Indic. 8, 691–703.

Davenport, J., Davenport, J.., 2006. the Ecology of Transportation: Managing Mobility for the Environment, environmental pollution. Springer, Dordrecht, The Netherlands.

Diaz-Varela, E.R., Vazquez-Gonzalez, I., Marey-Pérez, M.F., Álvarez-López, C.J., 2011. Assessing methods of mitigating wildlife–vehicle collisions by accident characterization and spatial analysis. Transp. Res. Part Transp. Environ. 16, 281–287.

Eastman, J.R., 2006. IDRISI. Worcester MA: Clark University.

Edman, T., Angelstam, P., Mikusiński, G., Roberge, J.-M., Sikora, A., 2011. Spatial planning for biodiversity conservation: Assessment of forest landscapes' conservation value using umbrella species requirements in Poland. Landsc. Urban Plan. 102, 16–23. doi:10.1016/j.landurbplan.2011.03.004

Eigenbrod, F., Hecnar, S.J., Fahrig, L., 2009. Quantifying the Road-Effect Zone: Threshold Effects of a Motorway on Anuran Populations in Ontario, Canada. Ecol. Soc. 14, 1–18.

Elith, J., Leathwick, J.R., 2009. Species Distribution Models: Ecological Explanation and Prediction Across Space and Time. Annu Rev Ecol Evol Syst 40, 677–97. doi:10.1146/annurev.ecolsys.110308.120159

ESRI, 2013. ArcGIS Desktop: Release 10. Environmental Systems Research Institute, Redlands, CA.

Etherington, T.R., 2012. Least-cost modelling on irregular landscape graphs. Landsc. Ecol. 27, 957–968. doi:10.1007/s10980-012-9747-y

Etherington, T.R., Holland, P.E., 2013. Least-cost path length versus accumulated-cost as connectivity measures. Landsc. Ecol. 28, 1223–1229. doi:10.1007/s10980-013-9880-2

Fahrig, L., Pedlar, J.H., Pope, S.E., Taylor, P.D., Wegner, J.F., 1995. Effect of road traffic on amphibian density. Biol. Conserv. 73, 177–182.

Fahrig, L., Rytwinski, T., 2009. Effects of Roads on Animal Abundance: an Empirical Review and Synthesis. Ecol. Soc. 14, 1–20.

Feurich, R., Boubée, J., Olsen, N.R.B., 2012. Improvement of fish passage in culverts using CFD. Ecol. Eng. 47, 1–8.

Findlay, T.C.S., Bourdages, J., 2000. Tiempo de Respuesta de la Biodiversidad de Humedales a la Construcción de Caminos en Tierras Adyacentes. Conserv. Biol. 14, 86–94.

Fleishman, E., Murphy, D.D., Brussard, P.F., 2000. A new method for selection of umbrella species for conservation planning. Ecol. Appl. 10, 569–579. doi:10.1890/1051-0761(2000)010[0569:ANMFSO]2.0.CO;2

Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., Holling, C.S., 2004. Regime Shifts, Resilience, and Biodiversity in Ecosystem Management. Annu. Rev. Ecol. Evol. Syst. 35, 557–581.

Forman, R.T.T., 1998. Road ecology: A solution for the giant embracing us. Landsc. Ecol. 13, III–V.

Forman, R.T.T., Deblinger, R.D., 2000. The Ecological Road-Effect Zone of a Massachusetts (U.S.A.) Suburban Highway. Conserv. Biol. 14, 36–46.

Forman, R.T.T., Godron, M., 1986. Landscape Ecology. Wiley.

Forman, R.T.T., Lauren, A.E., 1998. Roads and their major ecological effects. Annu Rev Ecol Syst 29, 207–231.

Forman, R.T.T., Sperling, D., Bissonette, J.T., Clevenger, A.P., Cutshall, C.D., Dale, V.H., Fahrig, L., France, R., Goldman, C.R., Heanue, K., Jones, J.A., Swanson, F.J., Turrentine, T., Winter, T.C., 2003. Road Ecology: Science and Solutions. Island Press.

Frair, J.L., Merrill, E.H., Beyer, H.L., Morales, J.M., 2008. Thresholds in landscape connectivity and mortality risks in response to growing road networks. J. Appl. Ecol. 45, 1504–1513.

Francis, C.D., Ortega, C.P., Cruz, A., 2011. Noise pollution filters bird communities based on vocal frequency. PloS ONE 6.

Freeman, C.L., 1979. Centrality in Social Networks Conecptual Clarification. Soc. Netw. 1, 215–239.

Freitas, S.R., Hawbaker, T.J., Metzger, J.P., 2010. Effects of roads, topography, and land use on forest cover dynamics in the Brazilian Atlantic Forest. For. Ecol. Manag. 259, 410–417.

Fu, W., Liu, S., Dong, S., 2010. Landscape pattern changes under the disturbance of road networks. Int. Conf. Ecol. Inform. Ecosyst. Conserv. ISEIS 2010 2, 859–867. doi:10.1016/j.proenv.2010.10.097

Geneletti, D., 2006. Some common shortcomings in the treatment of impacts of linear infrastructures on natural habitat. Environ. Impact Assess. Rev. 26, 257–267.

Geneletti, D., 2004. Using spatial indicators and value functions to assess ecosystem fragmentation caused by linear infrastructures. Int. J. Appl. Earth Obs. Geoinformation 5, 1–15.

Glasson, J., Therivel, R., Chadwick, A., 2001. Introduction to Environmental Impact Assessment, 2nd Edition. ed. Spon Press.

Glista, D.J., DeVault, T.L., DeWoody, J.A., 2009. A review of mitigation measures for reducing wildlife mortality on roadways. Landsc. Urban Plan. 91, 1–7.

Gontier, M., Balfors, B., Mörtberg, U., 2006. Biodiversity in environmental assessment--current practice and tools for prediction. Environ. Impact Assess. Rev. 26, 268–286.

Gontier, M., Mörtberg, U., Balfors, B., 2010. Comparing GIS-based habitat models for applications in EIA and SEA. Environ. Impact Assess. Rev. 30, 8–18.

Håkansson, M., Regazzoni, M., Lundberg, A.I., Gibrand, M., 2012. Feasibility study for railway link between Roslagsbanan and Arlanda Airport (Feasibility study No. SL-2011-05811). SL, Stockholm, Sweden.

Hanski, I., 1994. A Practical Model of Metapopulation Dynamics. J. Anim. Ecol. 63, 151–162.

Hanski, I., Gilpin, M., 1991. Metapopulation dynamics: brief history and conceptual domain. Biol. J. Linn. Soc. 42, 3–16.

Hanski, I., Ovaskainen, O., 2003. Metapopulation theory for fragmented landscapes. Theor. Popul. Biol. 64, 119–127.

Helldin, J.O., Seiler, A., 2003. Effects of roads on the abundance of birds in Swedish forests and farmland. EINE Conf. Proc.

Hepenstrick, D., Thiel, D., Holderegger, R., Gugerli, F., 2012. Genetic discontinuities in roe deer (Capreolus capreolus) coincide with fenced transportation infrastructure. Basic Appl. Ecol. 13, 631–638. doi:10.1016/j.baae.2012.08.009

Holderegger, R., Di Giulio, M., 2010. The genetic effects of roads: A review of empirical evidence. Basic Appl. Ecol. 11, 522–531.

Holling, C.S., 1973. Resilience and stability of ecological systems. Annu. Rev. Ecol. Syst. 4, 1–23.

Hoskin, C., Goosem, M.W., 2010. Road impacts on abundance, call traits, and body size of rainforest frogs in northeast australia. Ecol. Soc. 15.

Hovd, H., Skogen, A., 2005. Plant species in arable field margins and road verges of central Norway. Agric. Ecosyst. Environ. 110, 257–265.

Huang, I.B., Keisler, J., Linkov, I., 2011. Multi-criteria decision analysis in environmental sciences: Ten years of applications and trends. Sci. Total Environ. 409, 3578–3594. doi:10.1016/j.scitotenv.2011.06.022

Huijser, M.P., Bergers, P.J.M., 2000. The effect of roads and traffic on hedgehog (Erinaceus europaeus) populations. Biol. Conserv. 95, 111–116.

Hulme, P.E., 2009. Trade, transport and trouble: managing invasive species pathways in an era of globalization. J. Appl. Ecol. 46, 10–18.

Jackson, N.D., Fahrig, L., 2011. Relative effects of road mortality and decreased connectivity on population genetic diversity. Biol. Conserv. 144, 3143–3148.

Jaeger, J.A.G., Bowman, J., Brennan, J., Fahrig, L., Bert, D., Bouchard, J., Charbonneau, N., Frank, K., Gruber, B., von Toschanowitz, K., 2005. Predicting when animal populations are at risk from roads: an interactive model of road avoidance behavior. Ecol. Model. 185, 329–348.

Jaeger, J.A.G., Schwarz-von Rammer, H.G., Esswein, H., Muller, M., Schmidt-Luttman, M., 2007. Time Series of Landscape Fragmentation Caused by Transportation Infrastructure and Urban Development: a Case Study from Baden- Württemberg, Germany. Ecol. Soc. 12.

Janssen, R., 2001. On the use of multi-criteria analysis in environmental impact assessment in The Netherlands. J. Multi-Criteria Decis. Anal. 10, 101–109. doi:10.1002/mcda.293

Jay, S., Jones, C., Slinn, P., Wood, C., 2007. Environmental impact assessment: Retrospect and prospect. Environ. Impact Assess. Rev. 27, 287–300. doi:10.1016/j.eiar.2006.12.001

Jones, J.A., Swanson, F.J., Wemple, B.C., Snyder, K.U., 2000. Efectos de Carreteras en la Hidrología, Geomorfología y Parches de Perturbación en Redes de Arroyos. Conserv. Biol. 14, 76–85.

Kalantari, Z., Nickman, A., Lyon, S.W., Olofsson, B., Folkeson, L., 2014. A method for mapping flood hazard along roads. J. Environ. Manage. 133, 69–77. doi:10.1016/j.jenvman.2013.11.032

Karlson, M., Mörtberg, U., 2015. A spatial ecological assessment of fragmentation and disturbance effects of the Swedish road network. Landsc. Urban Plan. 134, 53–65. doi:10.1016/j.landurbplan.2014.10.009

Karlson, M., Mörtberg, U., Balfors, B., 2014. Road ecology in environmental impact assessment. Environ. Impact Assess. Rev. 48, 10–19.

Kiker, G.A., Bridges, T.S., Varghese, A., Seager, T.P., Linkov, I., 2005. Application of multicriteria decision analysis in environmental decision making. Integr. Environ. Assess. Manag. 1, 95–108. doi:10.1897/IEAM_2004a-015.1

Kvale, S., Brinkman, S., 2009. Interviews: Learning the craft och qualitative research interviewing, Second edition. ed. SAGE.

Laitila, J., Moilanen, A., 2013. Approximating the dispersal of multi-species ecological entities such as communities, ecosystems or habitat types. Ecol. Model. 259, 24–29. doi:10.1016/j.ecolmodel.2013.03.004

Lennartsson, T., Gylje, S., 2009. Infrastrukturens biotoper - en refug för biologisk mångfald. CBMs Skr. 31.

Malczewski, J., 1999. GIS and Multicriteria decision analysis. John Wiley & Sons, Inc., New York, Canada.

Manel, S., Holderegger, R., 2013. Ten years of landscape genetics. Trends Ecol. Evol. 28, 614–621. doi:10.1016/j.tree.2013.05.012

Marcantonio, M., Rocchini, D., Geri, F., Bacaro, G., Amici, V., 2013. Biodiversity, roads, & landscape fragmentation: Two Mediterranean cases. Appl. Geogr. 42, 63–72. doi:10.1016/j.apgeog.2013.05.001

Mata, C., Hervás, I., Herranz, J., Malo, J.E., Suárez, F., 2009. Seasonal changes in wildlife use of motorway crossing structures and their implication for monitoring programmes. Transp. Res. Part Transp. Environ. 14, 447–452.

McCall, S.C., McCarthy, M.A., van der Ree, R., Harper, M.J., Cesarini, S., Soanes, K., 2010. Evidence that a Highway Reduces Apparent Survival Rates of Squirrel Gliders. Ecol. Soc. 15.

McRae, B.H., 2006. Isolation by resistance. Evolution 60, 1551–1561.

McRae, B.H., Beier, P., 2007. Circuit theory predicts gene flow in plant and animal populations, in: Proceedings of the National Academy of Sciences of the USA 104:19885-19890.

McRae, B.H., Dickson, B.G., Keitt, T.H., Shah, V.B., 2008. Using circuit theory to model connectivity in ecology, evolution, and conservation. Ecology 89, 2712–2724.

Metzger, J.P., Martensen, A.C., Dixo, M., Bernacci, L.C., Ribeiro, M.C., Teixeira, A.M.G., Pardini, R., 2009. Time-lag in biological responses to landscape changes in a highly dynamic Atlantic forest region. Conserv. Issues Braz. Atl. For. 142, 1166–1177. doi:10.1016/j.biocon.2009.01.033

Ministry of Enterprise and Innovation, 2014. Appendice 2 to decision III 4 during cabinet meeting on April 3, 2014 concerning the development of the national transportation system for the period 2014-2025, Report no. N2014/1779/TE.

Morrison-Saunders, A., Retief, F., 2012. Walking the sustainability assessment talk — Progressing the practice of environmental impact assessment (EIA). Environ. Impact Assess. Rev. 36, 34–41. doi:10.1016/j.eiar.2012.04.001

Mörtberg, U., Balfors, B., Knol, W.C., 2007. Landscape ecological assessment: A tool for integrating biodiversity issues in strategic environmental assessment and planning. J. Environ. Manage. 82, 457–470.

Mörtberg, U., Zetterberg, A., Balfors, B., 2012. Urban Landscapes in transition - Lessons from Integrating Biodiversity and Habitat Modelling in planning. J. Environ. Assess. Policy Manag. 14, 1–31.

Mumme, R.L., Schoech, S.J., Woolfenden, G.E., Fitzpatrick, J.W., 2000. Life and death in the fast lane: demographic consequences of road mortality in the florida Scrub-Jay. Conserv. Biol. 14, 501–512.

Munck, I.A., Bennett, C.M., Camilli, K.S., Nowak, R.S., 2010. Long-term impact of de-icing salts on tree health in the Lake Tahoe Basin: Environmental influences and interactions with insects and diseases. For. Ecol. Manag. 260, 1218–1229. doi:doi: DOI: 10.1016/j.foreco.2010.07.015

Munda, G., Nijkamp, P., Rietveld, P., 1995. Qualitative multicriteria methods for fuzzy evaluation problems: An illustration of economic-ecological evaluation. Eur. J. Oper. Res. 82, 79–97. doi:10.1016/0377-2217(93)E0250-2

Munda, G., Nijkamp, P., Rietveld, P., 1994. Qualitative multicriteria evaluation for environmental management. Ecol. Econ. 10, 97–112. doi:10.1016/0921-8009(94)90002-7

Neel, M.C., 2008. Patch connectivity and genetic diversity conservation in the federally endangered and narrowly endemic plant species Astragalus albens (Fabaceae). Biol. Conserv. 141, 938–955.

Olsson, J., 2009. Improved road accessibility and indirect development effects: evidence from rural Philippines. J. Transp. Geogr. 17, 476–483.

Ottosson, M., Lennartsson, T., Svensson, R., 2012. Nya vägar till artrikedom. CBMs Skr. 66.

Pascual-Hortal, L., Saura, S., 2006. Comparison and development of new graph-based landscape connectivity indices: towards the prioritization of habitat patches and corridors for conservation. Landsc. Ecol. 21, 959–967.

Perdikaki, K., Mason, C.F., 1999. Impact of road run-off on receiving streams in eastern England. Water Res. 33, 1627–1633. doi:doi: 10.1016/S0043-1354(98)00396-0

Pereira, M., Segurado, P., Neves, N., 2011. Using spatial network structure in landscape management and planning: a case study with pond turtles. Landsc. Urban Plan. 100, 67–76.

Peters, D.P.C., Pielke, R.A., Bestelmayer, B.T., Allen, C.D., Munson-McGee, S., Havstad, K.M., 2004. Cross-scale interactions, nonlinearities, and forecasting catastrophic events. PNAS 101, 15130–15135.

Pinto, N., Keitt, T., 2009. Beyond the least-cost path: evaluating corridor redundancy using a graph-theoretic approach. Landsc. Ecol. 24, 253–266. doi:10.1007/s10980-008-9303-y

Pocock, Z., Lawrence, R.E., 2005. How far into a forest does the effect of a road extend? defining road edge effect in a eucalypt forests of South East Australia. Proc. 2005 Int. Conf. Ecol. Transp., Recent Work 397–405.

Reijnen, R., Foppen, R., Meeuwsen, H., 1996. The effects of traffic on the density of breeding birds in Dutch agricultural grasslands. Biol. Conserv. 75, 255–260.

Reijnen, R., Foppen, R., Veenbaas, G., 1995. disturbance by traffic of breeding birds: evaluation of the effect and considerations in planning and manging road corridors. Biodivers. Conserv. 6, 567–581.

Ribeiro, R., Carretero, M.A., Alarcos, N., Ortis - Santaliestra, M., Lizana, M., Llorente, G.A., 2011. The pond network: can structural connectivity reflect on (amphibian) biodiversity patterns? Landsc. Ecol. 26, 673–682.

Rodríguez - Pérez, J., García, D., Martínez, D., 2014. Spatial networks of fleshy-fruited trees drive the flow of avian seed dispersal through a landscape. Funct. Ecol. 28, 990–998.

Row, J.R., Blouin-Demers, G., Weatherhead, P.J., 2007. Demographic effects of road mortality in black ratsnakes (Elaphe obsoleta). Biol. Conserv. 137, 117–124. doi:10.1016/j.biocon.2007.01.020

Rubino, M.J., Hess, G.R., 2003. Planning open spaces for wildlife 2: modeling and verifying focal species habitat. Landsc. Urban Plan. 64, 89–104.

Rytwinski, T., Fahrig, L., 2012a. Do species life history traits explain population responses to roads? A meta-analysis. Biol. Conserv. 147, 87–98.

Rytwinski, T., Fahrig, L., 2012b. Do species life history traits explain population responses to roads? A meta-analysis. Biol. Conserv. 147, 87–98. doi:10.1016/j.biocon.2011.11.023

Rytwinski, T., van der Ree, R., Cunnington, G.M., Fahrig, L., Findlay, C.S., Houlahan, J., Jaeger, J.A.G., Soanes, K., van der Grift, E.A., 2015. Experimental study designs to improve the evaluation of road mitigation measures for wildlife. J. Environ. Manage. 154, 48–64. doi:10.1016/j.jenvman.2015.01.048

Saura, S., Pascual-Hortal, L., 2007. A new habitat availability index to integrate connectivity in landscape conservation planning: Comparison with existing indices and application to a case study. Landsc. Urban Plan. 83, 91–103. doi:10.1016/j.landurbplan.2007.03.005

Saura, S., Rubio, L., 2010. A common currency for the different ways in which patches and links can contribute to habitat availability and connectivity in the landscape. Ecography 33, 523–537.

Saura, S., Torné, J., 2009. Conefor Sensinode 2.2: A software package for quantifying the importance of habitat patches for landscape connectivity. Environ. Model. Softw. 24, 135–139. doi:10.1016/j.envsoft.2008.05.005

SCB, 2013. The landsuse of Transport Infrastructure.

Scheffer, M., Carptenter, S., Foley, J.A., Folke, C., Walker, B., 2001. Catastrophic shifts in ecosystems. Nature 413, 591.

Schindler, S., Poirazidis, K., Wrbka, T., 2008. Towards a core set of landscape metrics for biodiversity assessments: A case study from Dadia National Park, Greece. Ecol. Indic. 8, 502–514.

SEPA, 2012. Grön Infrastruktur (Skrivelse 2012-12-13 till M-departementet No. dnr NV-03013-12).

Shah, V.B., McRae, B.H., 2008. Circuitscape: A Tool for Landscape Ecology, in: Proceedings of the 7th Python in Science Conference (SciPy 2008).

Shepard, D.B., Kuhns, A.R., Dreslik, M.J., Phillips, C.A., 2008. Roads as barriers to animal movement in fragmented landscapes. Anim. Conserv. 11, 288–296.

Smith, A.B., 2010. Caution with curves: Caveats for using the species–area relationship in conservation. Biol. Conserv. 143, 555–564.

Söderman, T., 2009. Natura 2000 appropriate assessment: Shortcomings and improvements in Finnish practice. Environ. Impact Assess. Rev. 29, 79–86.

Söderman, T., 2006. Treatment of biodiversity issues in impact assessment of electricity power transmission lines: A Finnish case review. Environ. Impact Assess. Rev. 26, 319–338.

Spellerberg, I.F., 1998. Ecological Effects of Roads and Traffic: A Literature Review. Glob. Ecol. Biogeogr. Lett. 7, 317–333.

Sriyaraj, K., Shutes, R.B.., 2001. An assessment of the impact of motorway runoff on a pond, wetland and stream. Environ. Geochem. Trop. Subtrop. 26, 433–439. doi:10.1016/S0160-4120(01)00024-1

STA, 2012. Nationella Vägdatabasen, Trafikverket, Norrköping [National Road Database, Swedish Transport Agency]. Available online: http://www.nvdb.se/, accessed 2012-12-12.

Stohlgren, T.J., Schnase, J.L., 2006. Risk Analysis for Biological Hazards: What We Need to Know about Invasive Species. Risk Anal. 26, 163–173.

Summers, P.D., Cunnington, G.M., Fahrig, L., 2011. Are the negative effects of roads on breeding birds caused by traffic noise? J. Appl. Ecol. 48, 1527–1534.

Therivel, R., 2004. Strategic Evnironmental Assessment in Action. Earthscan.

Thompson, S., Treweek, J.R., Thurling, D.J., 1997. The Ecological Component of Environmental Impact Assessment: A Critical Review of British Environmental Statements. J. Environ. Plan. Manag. 40, 157–172.

Tischendorf, L., Fahrig, L., 2000. On the Usage and Measurement of Landscape Connectivity. Oikos 90, 7–19. doi:10.2307/3547105

Treweek, J., 1996. Ecology and evnironmental impact assessment. J. Appl. Ecol. 33, 191–199.

Trombulak, S.C., Frissell, C.A., 2000. Review of Ecological Effects of Roads on Terrestrial and Aquatic Communities. Conserv. Biol. 14, 18–30.

van der Ree, R., Gulle, N., Holland, K., van der Grift, E., Mata, C., Suarez, F., 2007. Overcoming the Barrier Effect of Roads-How Effective Are Mitigation Strategies?

van der Ree, R., Heinze, D., McCarthy, M., Mansergh, I., 2009. Wildlife Tunnel Enhances Population Viability. Ecol. Soc. 14.

van der Ree, R., Smith, D.J., Grilo, C., 2015. Handbook of Road Ecology. Wiley-Blackwell.

van Langevelde, F., van Dooremalen, C., Jaarsma, C.F., 2009. Traffic mortality and the role of minor roads. J. Environ. Manage. 90, 660–667. doi:10.1016/j.jenvman.2007.09.003

Vos, C.C., Verboom, J., Opdam, P.F.M., Ter Braak, C.J.F., 2001. Toward Ecologically Scaled Landscape Indices. Am. Nat. 157, 24–41.

Walz, U., Syrbe, R.U., 2013. Linking landscape structure and biodiversity. Link. Landsc. Struct. Biodivers. 31, 1–5. doi:10.1016/j.ecolind.2013.01.032

Yanes, M., Velasco, J.M., Suárez, F., 1995. Permeability of roads and railways to vertebrates: The importance of culverts. Biol. Conserv. 71, 217–222. doi:10.1016/0006-3207(94)00028-O

Yan, X., Zhenyu, L., Gregg, W.P., Dianmo, L., 2001. Invasive species in China - an overview. Biodivers. Conserv. 10, 1317–1341.

Zetterberg, A., Mörtberg, U.M., Balfors, B., 2010. Making graph theory operational for landscape ecological assessments, planning, and design. Landsc. Urban Plan. 95, 181–191.