Ecology, Transport Infrastructure and Environmental Assessment

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Licentiate thesis

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SUMMARY

Vägar och järnvägar har stor påverkan på djur, växter, vattenmiljöer och på landskapsbilden som helhet. Sett från ett bevarandeperspektiv är denna påverkan inte alltid negativ. Vissa djur och växter anpassar sig eller rent av trivs i väg och järnvägsmiljöer, vilket erbjuder en möjlighet att stärka biologiska och ekologiska värden i landskapet. Vägar och järnvägar kan också hindra eller accelerera viktiga ekologiska processer, såsom kolonisation och utdöende av populationer, migration och spridningmönster med kaskadeffekter på både högre och lägre trofinivåer. Sett ur ett större perspektiv är vägar och järnvägars påverkan övervägande negativ för biodiversiteten i stort, och byggnation, underhåll och bruk av vägar och järnvägar är idag starkare kopplat till förlust av biodiversitet än tidigare. I EU och följaktligen också i Sverige ska planering och byggnation av vägar och järnvägar miljöbedömas enligt Direktiv 85/337/EEC (projektnivå) och2001/42 (planeringsnivå). Dessa direktiv syftar till att förebygga onödig miljöpåverkan genom att identifiera, prediktera och utvärdera en plan, ett program eller ett projekts påverkan på bland annat biodiversitet. Denna licentiatavhandling sammanfattar två studier med inriktning mot miljöbedömning och vägar/järnvägars påverkan på biodiversitet. Den första studien analyserade hur biodiversitetsaspekter beaktades i miljöbedömningsprocesser av väg och järnvägsplaner och projekt. Miljöbedömningsdokument rörande väg och järnvägsplaner samt projekt granskades utifrån en lista med brister gällande beaktandet av biodiversitet, som sammanställdes utifrån vetenskapliga artiklar. Resultaten indikerade att hanteringen av biodiversitetsaspekter har förbättrats konitnuerligt. Resultaten indikerade dock betydande brister vad gäller hanteringen av fragmenteringseffekter, miljöbedömningsstudiens geografiska och temporära avgränsning samt att kvantitativa metoder inte användes för prediktion och utvärdering av påverkan på biodiversitet. Den andra studien analyserade det svenska vägnätets påverkan på grupper av djur och fåglararter, samt skllnader mellan fragmenteringseffekter och störningseffekter. Studien baserades på statistiska modeller presenterade i vetenskapliga artiklar, litteraturstudier och GIS-analyser. I studien indentifierades naturtyper och artgrupper som sannolikt är extra utsatta för väg eller järnvägseffekter. Sammanfattningsvis så finns det ett mycket starkt stöd i forskningen för att vägar och järnvägar har en signifikant påverkan på den omkringliggande naturmiljön, och att denna påverkan sträcker sig långt ut från vägen/järnvägen. Denna påverkan har till viss del hanterats inom ramen för miljöbedöminig. Men på grund av brister i gällande praxis, t.ex att de miljöbedömningsmetoder som används idag inte är kapabla att bedöma vissa effekter, så analyseras och bedöms inte en del allvarliga effekter, t.ex fragmentering. Vidare konstateras att dessa effekter skulle kunna analyseras och bedömmas med befintliga verktyg och metoder, samt att ett ökat användande av kvantitativa analysmetoder skulle komplettera och förbättra de analyser av vägar/järnvägars påverkan på biodiversitet som görs idag.

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Mårten Karlson, May 2013

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List of papers

I. Karlson M., Mörtberg U. and Balfors B. Road ecology in environmental impact assessment. *Manuscript, submitted to Environmental Impact Assessment Review.*

II. Karlson M., Mörtberg U. and Balfors B. A spatial assessment of fragmentation and disturbance effects of the Swedish road network. *Manuscript.*

ABSTRACT

Transport infrastructure has a wide array of effects on ecological processes. These effects benefit certain species and might enhance or accelerate ecological processes such as colonization and dispersal, but as well extinction. The overall impact on biodiversity is however negative and several authors conclude transport infrastructure to have detrimental effects on terrestrial and aquatic communities. Planning and construction of transport infrastructure is in the EU to be preceded by an environmental assessment process, with the overall aim to prevent rather than repair potential unintended negative effects. This thesis presents two studies on transport infrastructure effects on biodiversity in the context of environmental assessment. The first study reviewed how and how sufficiently biodiversity aspects were accounted for in environmental assessment of transport infrastructure projects and plans, and identified opportunities to improve concurrent practice. The first study concluded that the treatment of biodiversity aspects has improved over the years, but that the low use of quantitative impact assessment methods, the treatment of fragmentation and spatial and temporal delimitation of the impact assessment study area remain problematic. The second study assessed the impact of the Swedish road network on biodiversity by use of existing landscape ecological metrics and GIS. The second study reconnects to the shortcomings in environmental assessment practice identified in the first study, by discussing the utility of the method in terms of applicability in environmental assessment processes. The second study identified nature types and species adversely exposed to transport infrastructure effects, and concluded that sound methodologies for biodiversity assessment can be developed using existing tools and techniques. In sum, transport infrastructure influence vast areas of the surrounding landscape, and this is not accounted for in planning and design of new transport infrastructure due to shortcomings in current environmental assessment practice. Existing tools and techniques could be used to address several of these shortcomings, and an increased use of quantitative analysis of transport infrastructure effects on biodiversity would add significantly to the quality of impact predictions and evaluations.

Key words: Environmental Impact Assessment; Strategic Environmental Assessment; Ecological Processes; Biodiversity; Transport Infrastructure; Roads

1. INTRODUCTION

The construction, utilization and management of transport infrastructure have shown to introduce a wide range of changes in terrestrial and aquatic environments, resulting in both positive and negative consequences for biodiversity (Spellerberg 1998; Trombulak and Frissell 2000; Forman et al. 2003). New constructions typically imply habitat loss, but as well new suitable habitat for some species (Fahrig and Rytwinski 2009; Lennartsson and Gylje 2009). In addition, linear constructions implicitly sub-divide its surroundings, potentially introducing barriers to movement with implications for ecological processes such as colonization and extinction, species distribution and genetic exchange (Hanski and Ovaskainen 2003; Seiler et al. 2003; Holderegger and Di Giulio 2010; Jackson and Fahrig 2011). Roads and railways alter hydrological patterns due to their physical structure, with subsequent effects on erosion and sedimentation rates and may act as barriers to animal movement or as dispersal conduits (Forman et al. 2003; Gelbard and Belnap 2003). Utilization of transport infrastructure further implies disturbance like traffic noise and artificial light, increased animal mortality, introduction of exotic species and chemical contamination. These impacts extend outwards from the road corridor creating effect zones within which environmental conditions are altered by among other things, the presence of non-endemic or exotic flora, reduced species richness and abundance, altered micro-climatic condition and by human induced disturbance (Forman and Alexander 1998; Spellerberg 1998; Trombulak and Frissell 2000). Substantial amounts of empirical research support the existence of effect zones (e.g., Forman and Deblinger 2000; Biglin and Dupigny-Giroux 2006; Eigenbrod et al. 2009; Bissonette and Rosa 2009; Benétez-Lópezet al. 2010), and studies have estimated 15-20 % of the US (Forman and Alexander 1998) and 16 % of the Netherlands (Reijnen et al. 1995) to be within effect zones. However, the cause for the observed effects on species richness and abundance is still debated. Species traits and attributes are likely to influence how they coop with transport infrastructure (Rytwinski and Fahrig 2012). Traffic noise have been proposed to reduce species richness and abundance of birds and amphibians (Reijnen et al. 1996; Forman and Deblinger 2000; Helldin and Seiler 2003; Eigenbrod, et al. 2009). Road mortality is by some authors argued to be a major cause for reduced abundance and diversity (Mumme et al. 2000; Summers et al. 2011). Additionally, the reduced species richness and abundance can as well be explained by meta-population theory, where habitat amount and quality and the processes of migration, colonization and extinction determine the viability of populations and genetic exchange. A meta-population perspective would suggest habitat loss, fragmentation and barrier effects to be primary drivers for reduced species richness and abundance (Hanski and Gilpin 1991; Hanski and Ovaskainen 2003; Jaeger et al. 2007). In Sweden and the EU, construction of transportation infrastructure is to be preceded by an Environmental Impact Assessment process (an EIA-process) according to Directive 85/337/EEC concerning projects, and transport infrastructure plans and programs are to be preceded by a Strategic Environmental Assessment process (an SEA-process) according to Directive 2001/42. In 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), used as a basis for decision making and to inform the general public (Glasson et al. 2001; Therivel 2004). The treatment of biodiversity impacts in EIA and SEA on transport infrastructure projects has received much criticism over the years; for not fulfilling the requirements of the Directives 85/337/EEC and 2001/42, and for neglecting the full extent of the influence transport infrastructure has on biodiversity (Treweek 1996; Thompson et al. 1997; Atkinson et al. 2000; Gontier et al. 2006; Geneletti 2006). Despite advances in methods for prediction of ecological impacts, assessments of impacts on biodiversity are done by qualitative analysis and consultations.

1.1. Aim and objectives

The aim of this thesis was to study how transport infrastructure interacts with ecological processes and to review the state of treatment of these interactions in environmental assessment processes, in order to identify windows of opportunity for improving biodiversity considerations in EIA and SEA. The specific aim of Paper I was to analyze how biodiversity has been considered in environmental assessment processes on transport infrastructure projects. This consisted of a literature review and by reviewing environmental impact statements and environmental reports. The specific aim of Paper II was to conduct a spatial assessment of how the Swedish road network impacts on Swedish birds and mammals by preforming spatial modeling of different transport infrastructure effects reported in the literature. The aim of Paper

II also involved to compare transport infrastructure impacts according to different perspectives on cause effect relationships, and to evaluate the utility of the methods used for environmental assessment purposes.

1.2. Structure of the thesis

As the research presented in this thesis has a theoretical background in the three disciplines of systems ecology, landscape ecology and a sub-discipline to physical planning – environmental assessment, a brief explanation to these research areas is given in Chapter 2. Chapter 3 describes the methods used in Paper I and Paper II. The terms "sustainable development" and "biodiversity" are frequently used in this thesis, therefore an explanation to the importance of biodiversity for sustainable development is included in the "state of the art" in Chapter 4. Concluding remarks are presented in Chapter 5, and Chapter 6 presents some ideas for future research.

2. THEORETICAL FRAMEWORK

2.1. Complex adaptive systems

Research on complex adaptive systems (CAS) focuses on the system dynamics, or the system behavior. Traditionally, ecosystems have been considered to be either in a successional stage in which species with different competitive capacity colonize the system, to eventually become dominated by the strongest competitor species and reach an equilibrium stage, where it would stabilize and accumulate biomass and energy e.g. trees eventually building up a forest (Holling and Meffe 1996; Resillience Alliance 2013). This perspective has been the fundament for most conservational management strategies, and has inspired management to maintain stability and control even naturally occurring disturbances, sometimes resulting in adverse unintended consequences (Holling and Meffe 1996). The theory of complex adaptive systems suggests this traditional view to be complemented with two additional stages in ecosystem dynamics, namely a release stage following the conservation stage and reorganization phase preceding the successional stage (Holling 2004). These four stages are referred to as the "Adaptive Cycle" (Fig. 1), and the dynamics of CAS is typically characterized by several adaptive cycles which are nested across space and time (Fig. 2). The term "panarchy" was coined to describe the order of interaction between adaptive cycles in CAS where small, large, slow and fast moving cycles interact and exert mutual feedback on each other across scales (Holling and Gunderson 2002), as in contrast to a hierarchical order of interaction where dominating cycles e.g. large slow moving, would influence small fast moving ones . A significant difference between perceiving ecosystem dynamics as panarchic as opposed to hierarchic, is the inherent and eternal change emphasized in the panarchic view. In the panarchic view, the equilibrium stage will collapse eventually and the system will need to reorganize, either into the previous ecosystem but potentially into a different ecosystem (Holling 2004; Gotts 2007). Complex adaptive systems theory is a theoretical framework that helps interpret ecosystem phenomena that do not fit into the growth and conservation phase. It has led to new insights in natural resource management (Folke et al. 2002), but have just begun to influence planning and environmental assessment (Slootweg and Jones 2011).

2.2. Landscape ecology

Landscape Ecology is the study of how biophysical patters shape biotic and abiotic processes in a given landscape, and how these processes in return shape landscape patterns. The term "Landscape Ecology" was first mentioned by Carl Troll, who had specialized in plant geography, in 1939 in a study of the application of aerial photography's showing vegetation densities. Troll later defined landscape ecology as "the study of the entire complex cause-effect network between communities of species and their environmental conditions within a given landscape" (Bastian et al. 2002). Landscape ecological research

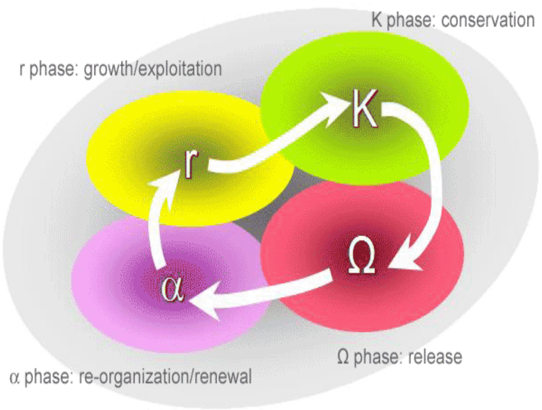

Figure 1 The four phases of ecosystem dynamics. Source: www.resalliance.org

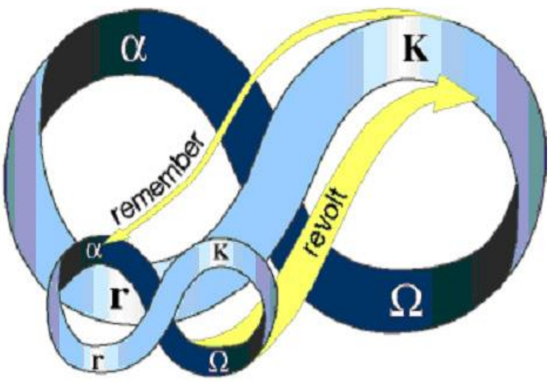

Figure 2 Interaction between adaptive cycles across scales (Holling 2004)

consists today of several schools, of which at least two can be distinguished. The work of Carl Troll developed into the European school, where humans are considered as a biophysical driver that shape landscape patterns, central for understanding landscape function. This anthropocentric perspective has brought the European school close to application in management and planning. The North American school was founded in the 1980s, primarily based on the on spatial pattern framework and the book on landscape ecology published by Forman and Godron (1986). The North American school differs from the European school by being less strictly concerned about the spatial extent of the "landscape" as a study area of pattern-process relationships, and by putting a stronger emphasis on the influence of spatial heterogeneity on processes, and less emphasis the role of anthropocentric activities (Turner 2005). Central to the study of landscape patterns and processes is also meta-population theories (Hanski and Gilpin 1991). Meta-population theories concern the ecological processes of colonization and extinction, and how these are determined by landscape patterns such as connectivity, size and amount of habitat in a patch mosaic landscape. Overall, landcape ecology comprises several tools and methods suitable for physical planning and environmental assessment.

2.3. Environmental assessment

Directive 85/337 on the assessment of the effects of certain public and private projects on the environment, and Directive 2001/42, on the assessment of the effects of certain plans and programs on the environment are two legislative tools 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 precedent 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. Environmental Impact Assessment (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 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 was added to Annex 1 and 2. In the end of 2011, the EIA-directive and its three amendments was codified into directive 2011/92/EU. 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 were seldom room making changes to an activity in its project phase. This emphasized the need for environmental assessment already at 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 (Glasson et al. 2001; Therivel 2004). The research presented in this thesis focuses on how and how sufficiently the requirements laid down in the EIA and SEA Directives consider biodiversity aspects in projects and plans.

3. METHODS

3.1. Literature review

In Paper I, a literature review was conducted on the effects of road and railway networks on landscape patterns and ecological processes. Further, literature on the treatment of biodiversity in Environmental Impact Statements (EIS) and Environmental Reports (ERs) was reviewed, and problem categories were identified by means of a content analysis according to Kvale and Brinkman (2009). In Paper II, research articles on transport infrastructure effects on ecological processes were identified and reviewed with the purpose to identify results and conclusions suitable for spatial modeling. In Paper II, a second literature review was carried out on the ecology and ranging behavior of Swedish bird and mammal species with the purpose to develop a set of ecological profiles to use as response variables in GIS analysis (Vos et al. 2001; Angelstam et al. 2004; Mörtberg et al. 2012).

3.2. Review of environmental assessment reports

EIS/ERs produced in Sweden, Scotland, Wales and the UK between 2005 and 2012 were reviewed using the problem categories, identified in the literature review, as checklist. Of the 16 EIS/ERs, 6 were ERs and 10 were EISs, of which 10 concerned road projects, 2 concerned railway projects and 4 were plans concerning both modes of transportation.

3.3. Spatial national assessment

3.3.1. The study area

Two different study areas were used in Paper II (Fig. 3). For the national assessment, the whole of Sweden was studied. Sweden is an elongated country comprising three different climatic regions (sub-arctic, cold-temperate and warm-temperate) measuring 1572 km from north to south (Nationalencyclopedin 2013; SMHI 2013). A rough description of the present land cover would be 50 % forest, 15 % agricultural land, 10 % water and around 3 % urban areas (Statistics Sweden 2008). For the disturbance and fragmentation analysis, the counties of Stockholm, Uppsala and Västmanland in their entirety, and parts of another 5 adjacent counties constituted the study area with a total extent of 43 827 km^2. This study area was characterized by vast areas of boreal forest, transcending eastwards towards the Baltic Sea into a mixture of green and blue areas and finally into coastal archipelago. The study area comprises areas with high biodiversity as well as rapidly urbanizing regions and sparsely populated rural areas.

3.3.2. The effects of infrastructure on birds and mammals

The results presented by Benétez-López et al. (2010) on the effects of infrastructure on birds and mammals was used to model the ecological impacts of the Swedish road network. The data used for the meta-analysis in this study was scrutinized with the purpose to exclude data from studies performed in environments non-similar to Swedish conditions. Two geo-spatial

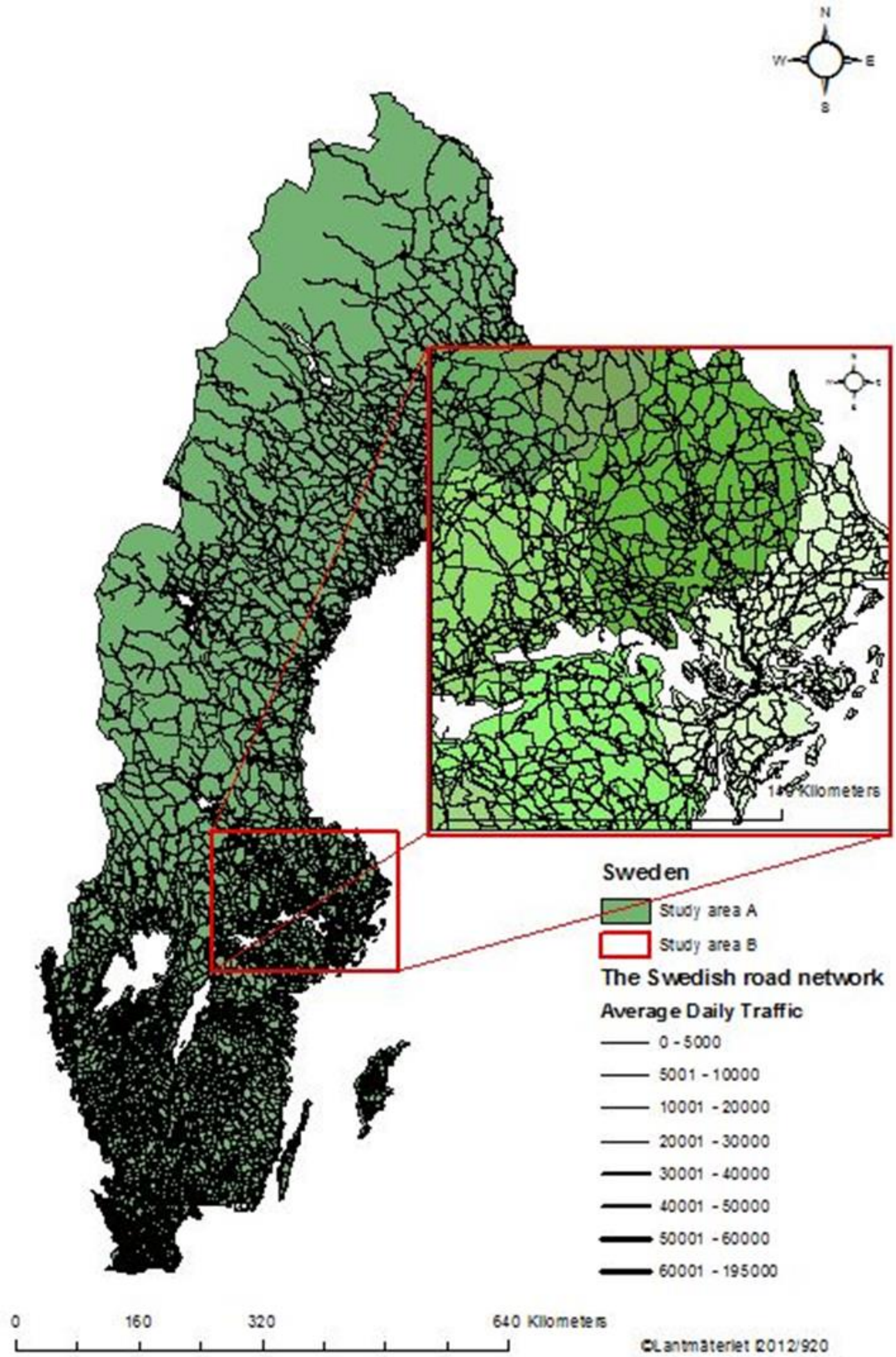

Figure 3 Study area A and B in used for the analysis in Paper II

datasets of the state-road network (Trafikverket 2012) were created, one for birds with an Average Daily Traffic (ADT) of >1000 and one for mammals with and ADT of >100. Six geo-spatial datasets, (four maps showing upper and lower 95% Confidence Intervals (CI), showing the predicted Mean Species Abundance (MSA) as a function of distance to infrastructure were created from the two meta-regression models in Benétez-López et al. (2010) by applying logit transformation in ArcMap (ESRI 2011) :

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

Where $\boldsymbol{MSA_{(estimated)}}$ is the predicted Mean Species Abundance at the observed distance from the road network ranging from 0-1, $\boldsymbol{u}$ is the linear equation describing the relationship between MSA and distance to infrastructure:

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

where a = the estimated value of u where $x = 0$, b = the regression coefficient for the independent variable x. In this study, x was each cell in a raster containing the Euclidian distance from a road, calculated from the birds' and the mammals' road-datasets. The MSA-datasets were then reclassified into four effect intensity zones (Table 1), overlaid on each of five datasets showing the spatial distribution of spruce, pine, hardwood deciduous forest, trivial deciduous forest and grasslands of high conservational value (Naturvårdsverket 2012), and the area overlaid by each zone calculated. CI intervals were calculated as the difference between the MSA and the upper and lower CI datasets. All GIS-analyses were performed in ArcGIS (ESRI 2011) at a resolution of 25x25 m.

3.3.3. Ecological profiles

Geo-spatial dataset in the form of a habitat network were created in ArcGIS (ESRI 2011) for each of the six ecological profiles representing forest and grassland-dwelling birds and mammals with high and low resource requirements. The habitat networks were created by reclassifying the Swedish landcover data (Lantmäteriet 2012) and nature types of high conservation value (Naturvårdsverket 2012) into habitat suitability maps. Habitat patches were then generated using focal statistics assuming "a patch" to be all available pixels within the daily activity range. Further, territories were assumed to be of circular shape, daily activity ranges used as radii, and pixels with suitability <0.5 were excluded from the datasets. Assumptions on habitat suitability and radii used are presented in Table 2 of Paper II in this thesis. For the grassland birds profile, 3 habitat networks were created due to the scarce availability of data on ranging behavior. In sum, 13 habitat networks were created. The GIS-analyses were performed at a resolution of 50 x 50 m.

Table 1 Effect intensity zones ranging from 0-1.0

MSA intervals				
Mammals	0.35-0.5	0.5-0.7	0.7-0.9	0.9-1.0
Birds	0.3-0.5	0.5-0.7	0.7-0.9	0.9-1.0

3.3.4. Comparing ecological impacts of fragmentation and disturbance

Disturbance and fragmentation effects were assessed by quantification of the structural properties of the ecoprofile networks. Network properties were quantified by calculating three landscape ecological metrics. A comparison was performed by evaluating the differences in these metrics between "fragmentation" of an ecoprofile and "disturbance" of the same ecoprofile. The comparison was essentially based on the hypothesis that disturbance effects to some extent might alter the physical properties of an ecoprofile network by altering its quality. Fig. 4 shows an explanatory workflow of the comparative analysis. A total of 13 ecoprofile networks were created after overlaying the habitat suitability maps with a geospatial dataset of the state road network (Trafikverket 2012) and excluding intersected pixels (fragmentation effects), and another 13 ecoprofile networks were created after overlaying the habitat suitability maps with the geospatial datasets on the effect intensity zones constructed earlier (disturbance effects). Zones with an MSA of < 0.5 were overlaid, the intersected pixels were reclassified into half the previous suitability, and new habitat networks were created. The GIS-analysis was performed at a resolution of 50 x 50 m, thus roads were considered to be 50 m wide. FRAGSTATS (McGarigal et al. 2012) was then used to calculate three class level metrics describing the structural properties of the habitat networks (Table 2). Upper and lower CI was calculated as the difference in metrics between the radii used for each profile.

Table 2 Description of metrics calculated

Landscape metrics			
Name	Total Class Area (CA)	No. of patches (NP)	% of study area (PLAND)
Description	Total area of all habitat patches in Ha	No. of individual patches	Class area as percentage of total area

4. RESULTS AND DISCUSSION

4.1. State of the art

4.1.1. The role of biodiversity in sustainable development

In ecology, empirical observations and theoretical experiments of the dynamics of ecosystems are strengthening consensus on the proposition that ecosystems can exist and operate in multiple stable states, and that shifts between states occur suddenly, often triggered by stochastic events after long time periods of gradual change, stress or anthropogenic disturbance (Scheffer et al. 2001; Scheffer and Carpenter 2003; Peters et al. 2004; Folke et al. 2004; Anderies, Walker, and Kinzig 2006; Rockström et al. 2009). Ecosystems are further proposed to be complex adaptive systems, and ecosystem dynamics are described in terms of adaptive cycles nested across space and time, in which ecosystems continuously transcend between four phases and that eventually, the system will collapse and reorganize either into the previous configuration or into a novel state (Holling 2004). In some ecosystems, shifts between states have been associated with loss of Ecosystem Services (ES) (Walker 1993; Scheffer et al. 2001; Folke et al. 2004; B. Walker and Salt 2006a). Biodiversity is of key importance for the capacity of the system to reorganize into the precedent configuration and consequently for the system to continue to produce ES. Biodiversity offers redundancy and diversity of ecosystem functions (species and communities), and diversity of responses to change and disturbance (different species coop with change differently even though active in the same niche), reducing the risk of losing crucial ecosystem functions in case of detrimental events and strengthens its capacity to reorganize and maintain its functions (Peterson et al. 1998; Folke et al. 2004). The capacity of an ecosystem to reorganize into its previous configuration after disturbance is referred to the ecosystems "resilience". Resilience as a property of ecosystems was first coined in Holling (1973) to explain dynamics of natural systems that stabilized and remained operational in dramatically different states, and which alternated between states in stable cycles. Resilience is today, in an ecological context, often defined as "the ability of a system to absorb disturbance and return to its original state" even though several other definitions exists (Brand and Jax 2007). Thus resilience has come to encompass the role of biodiversity in the dynamics of complex adaptive systems. Sustainable development, defined in the Brundtland Report (1987) as "a development that meets the needs of the present without compromising the abilities of future generations to meet their own needs", presumes todays' consumption of ES to still be there in the future. Transport infrastructure could threaten ecosystem resilience if negative effects are not identified and subsequently mitigated. However, transport infrastructure offers as well several opportunities to enhance ecosystem resilience as transport infrastructure constitute suitable habitat for various plant and insect species (Lennartsson and Gylje 2009).

4.1.2. Transport infrastructure and ecological processes

The effects of transportation infrastructure on ecosystems has been concluded to be overall change-inducing in several studies (Forman and Lauren 1998; Spellerberg 1998; Trombulak and Frissell 2000; Forman et al. 2003; Fahrig and Rytwinski 2009; Benétez-López et al. 2010; Rytwinski and Fahrig 2012) and its effects can fundamentally be divided into 3 categories 1) effects from removal and fragmentation of habitat during construction 2) effects of its physical structure and novelty as a landscape element; by altering hydrological processes and introducing barriers and dispersal conduits for species and 3) effects from its utilization; degradation of habitat quality through disturbance, pollution, introduction of exotic species and through increased species mortality. Additionally, transport infrastructure is considered to provide socio-economic incentives for alternative land uses (Jaeger et al. 2007; Freitas et al. 2010), and land use change is considered to be the major driver for loss of ecosystem

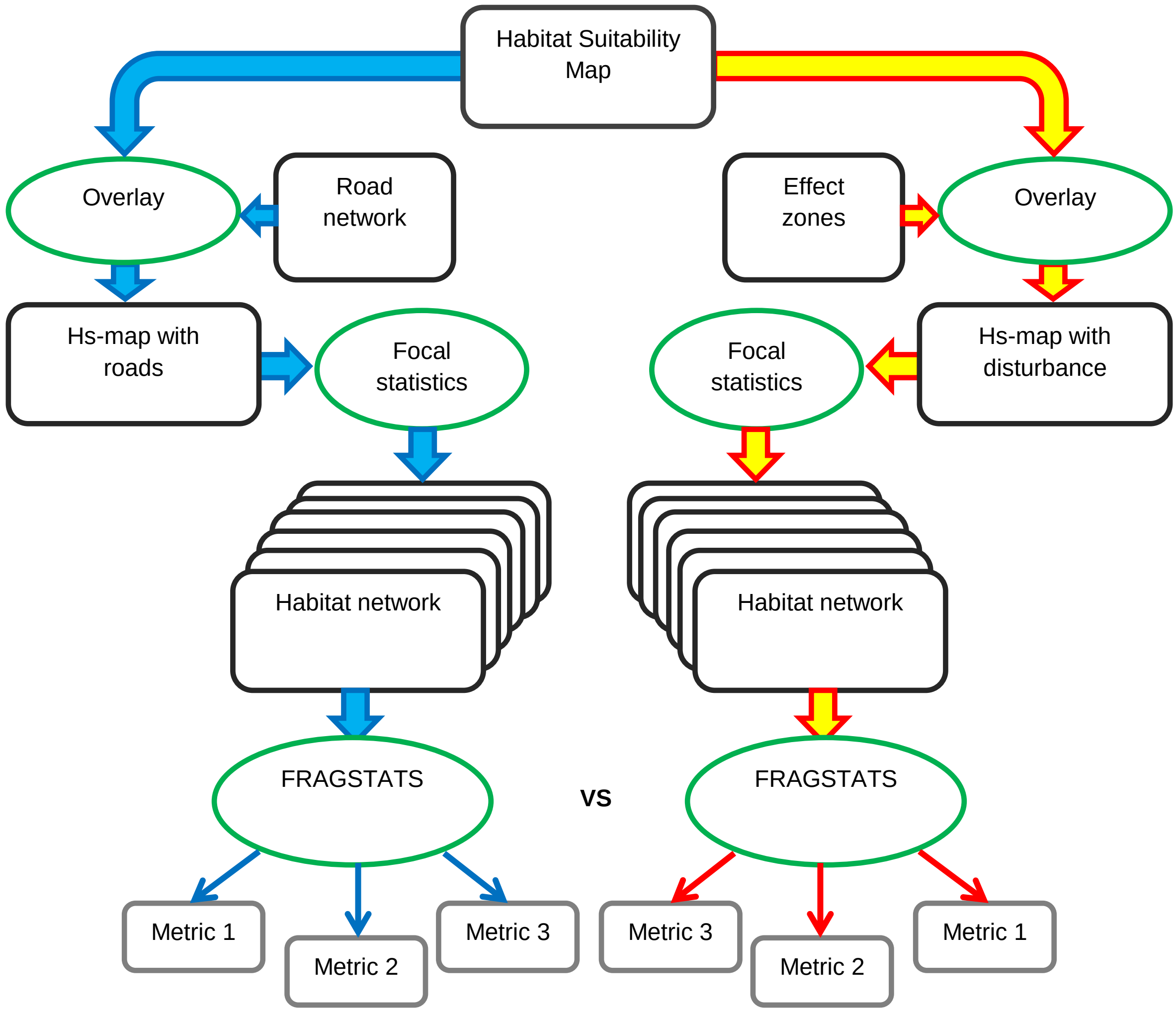

Figure 4 Workflow for the comparative analysis between fragmentation and disturbance effects. Habitat suitability maps for each of 6 ecoprofiles were overlaid by a road dataset and an effect zone dataset. Pixels overlaid by the road dataset (fragmentation effects) were excluded, and pixels overlaid by the effect zone dataset (disturbance effects) were reclassified into half the previous suitability. Two habitat networks were then created using focal statistics for each ecoprofile, and their spatial properties were analyzed using FRAGSTATS

functioning and services globally (Rockström et al. 2009). Several plant and animal species benefit from transport infrastructure environments and transportation networks offer several opportunities to strengthen biodiversity, but in sum, transportation networks have overall negative effects on biodiversity (Fahrig and Rytwinski 2009) and the effects have been suggested to extend up to several kilometers from the road (Forman and Deblinger 2000; Ritters and Wickham 2003; Biglin and Dupigny-Giroux 2006; Benétez-López et al. 2010) which is also indicated from the results of Paper II in this thesis. Transport infrastructure effects on the environment has during the last decade developed into an new branch of landscape ecology (Forman et al. 2003; Davenport and Davenport 2006).

4.1.3. The treatment of biodiversity in EIA and SEA

In Sweden as well as in the rest of the EU, Directive 2011/92/EU on "the assessment of the effects of certain public and private projects on the environment" (the EIA Directive) and Directive 2001/42 on "the assessment of the effects of certain plans and programs on the environment "(the SEA Directive) both require environmental assessment to be performed prior to planning and construction of new roads.

These directives call for a process in which significant environmental impacts are to be identified, assessed and evaluated and stakeholders informed and consulted, and that these aspects might influence the design of the plan or project (Glasson et al 2001; Therivel 2004). The overall purpose of these directives is to promote sustainable development. Paragraph 2 in Directive 2011/92/EU states that "*Effects on the environment should be taken into account at the earliest possible stage in all the technical planning and decision-making processes*". Informational requirements on impacts relating to biodiversity are mentioned in paragraph 14 "*…to ensure maintenance of the diversity of species and to maintain the reproductive capacity of the ecosystem as a basic resource for life."* and in paragraph a-d of article 3, "*fauna and flora, soil, water, air, climate and the landscape*" and "*the interaction between the factors referred to…*". Corresponding statements in Directive 2001/42 are found in paragraph 3 "*…to integrate as far as possible and as appropriate the conservation and sustainable use of biological diversity."*, Article 1 "*The objective of this directive is to provide for a high level of protection of the environment…with a view to promoting sustainable development…"* and in paragraph f of Annex 1 to Article 5 "*…issues such as biodiversity,…fauna, flora, soil, water, air, climatic factors,…and the interrelationship between the above factors".* However, several authors have suggested that impacts on biodiversity are still not adequately assessed in planning and construction of new transport infrastructure (Fig. 5; Fig. 1 and 2 in Paper I; Geneletti 2006; Gontier et al. 2006 among others) and concerns about the current application of EIA being insufficient in protection of biodiversity (COWI 2009) partly contributed to a recent revision of the recently codified EIA-directive 2011/92/EU (EC 2012). Karlson et al. (2013) suggest that todays' EIAs and SEAs on transport infrastructure projects and plans are executed in good compliance with the directives, but that some inadequacies in the treatment of biodiversity persist (Fig. 5). Karlson et al. (2013) point out that remaining problems concern the spatial and temporal delimitation of the impact assessment study area which in many cases could be increased, that ecological impacts remain descriptive and not quantified even though quantitative methods exists (for an overview, se e.g., Gontier et al. (2006;2010)), and the specific topic of fragmentation, which is often mentioned but its effects seldom analyzed. The concept of resilience is sometimes argued to be the key to sustainable development (Walker and Salt 2006b) has also recently been brought up on EIA and SEA-forums. Reflections on resilience and environmental assessment were summarized in Slootweg and Jones (2011), and a case-study on resilience an SEA was presented in Faith-Ell and Heikki (2011).

4.2. Effects of the Swedish road network

4.2.1. Summary of results

The results from the spatial assessment of fragmentation and disturbance showed that grasslands and hardwood deciduous forests were proportionally more exposed to road effects than spruce, pine and trivial deciduous forest (Fig. 6; Fig. 4 and 5 in Paper II). Of the six ecoprofiles used as response variables to fragmentation and disturbance effects, the high demands forest profile gave the greatest response to both effects, with a relative decrease in both CA and NP of more than 0.5 (Fig. 7 and 9). Disturbance reduced habitat amount (CA) and number of patches (NP) in all ecoprofiles except for the low demands forest profile, where NP increased as a response to disturbance (Fig. 9). Fragmentation also reduced CA in all profiles, but increased NP in the grassland ecoprofiles with high and low resource demands (Fig. 10). Both bird ecoprofiles responded stronger to disturbance than to fragmentation, with grassland birds showing a slightly greater response to both effects (Fig. 7-10). However, available habitat for grassland birds constituted 6-8% of the study area, compared to ca. 1% for forest birds (paper II, Fig. 14 and 15). In summary, grasslands and hard wood deciduous forests were more intensely exposed to road effects

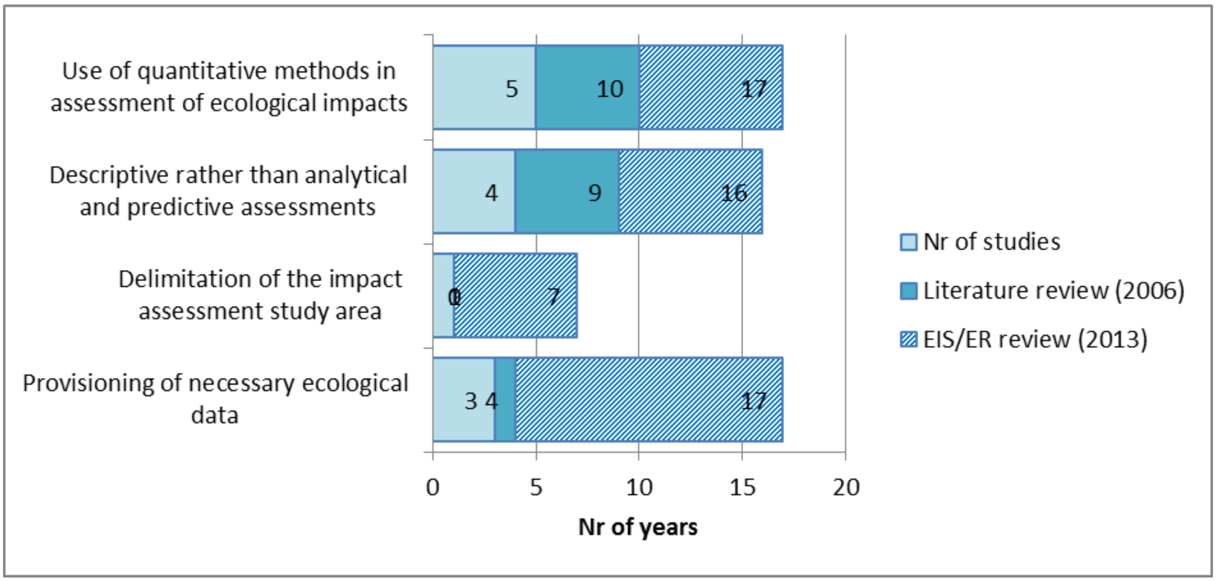

Figure 5 Contemporary problems with the treatment of biodiversity in EIA and SEA on transport infrastructure projects. The X-axis show the number of years that have passed between the first and the last time a problem category was appreciated in a research article relative to 1) the literature review and 2) the EAR-review. The x-axis also show the number of articles that has highlighted a problem category

than spruce, pine and trivial deciduous forest. The results also suggest fragmentation effects to be a greater concern for grassland species compared to forest species, and habitat loss to be a specific concern for forest species with high resource demands, and a general concern for mammal and bird species of both habitat types.

The degree of sub-division and total area of the ecoprofile networks depends to a large degree on the radii used, with small radii producing more fragmented networks with small patches and large radii producing more spatially aggregated networks with larger patches. Thus, a spatially aggregated network with large patches is likely to be relatively more fragmented and disturbed than an already highly fragmented network with small patches overlaid by one and the same fragmentation or disturbance data. From the perspective that large activity ranges (radii) are due to high resources demands or resource scarcity the response of the high demands forest profile was realistic, and could be interpreted as how the capacity of such habitat to support forest mammals with high resource demands was reduced by transport infrastructure effects. However, such a relationship between radii and level of response in the metrics was not identified in every ecoprofile, and it is likely that the spatial properties of the habitat suitability maps used in creation of the ecoprofiles also determine an ecoprofiles' sensitivity to fragmentation and disturbance effects.

4.2.2. Applicability in environmental assessment

The scarce use of quantitative methods in impact assessment of biodiversity has received much criticism over the last decade (Treweek 1996; Geneletti 2006; Gontier et al. 2006 among others), and in the EIS/ER-review undertaken in Paper I, qualitative matrices was the sole method used for assessment of biodiversity related impacts. Further, the treatment of fragmentation – an explicit impact of transport infrastructure projects, has also been criticized by these and other authors. Additionally, although fragmentation effects were mentioned in more than half of the EIS/ERs reviewed in Paper I, they were not analyzed in any. The method used in Paper II was based on the idea of that impacts on the functional properties of habitat networks can assessed by measuring relative changes in structural properties as response to transport

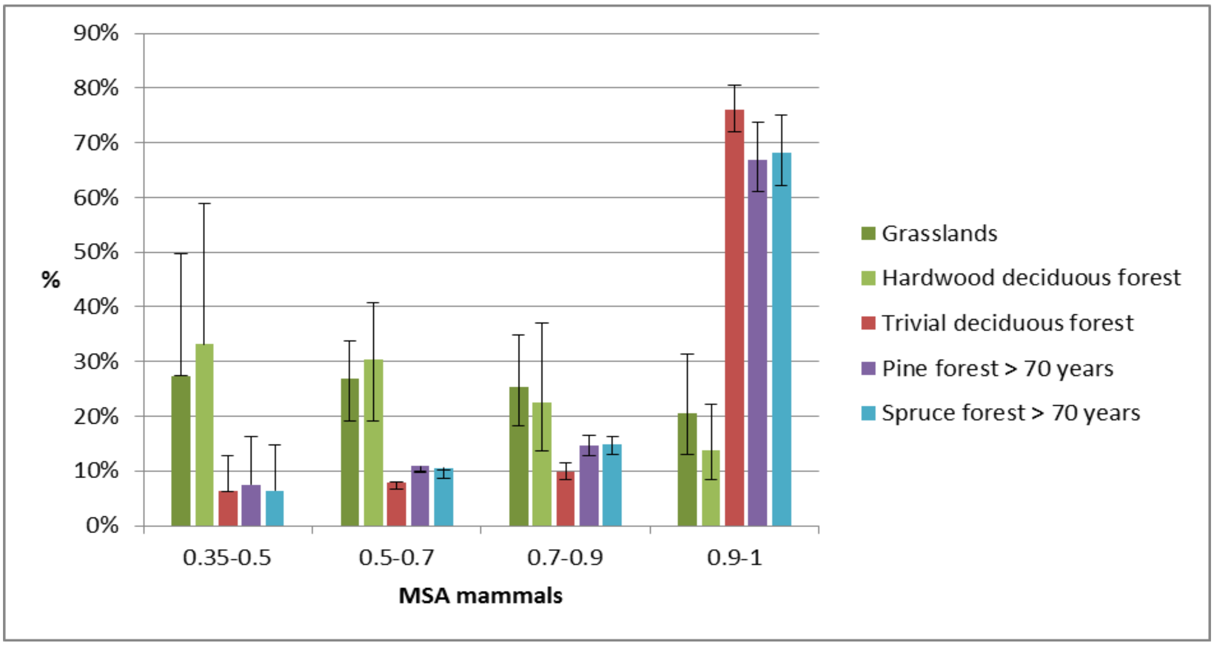

***Figure 6** Proportional share of highly valuable nature types per interval of road effect intensity expressed as reduced Mean Species Abundance of mammals*

infrastructure effects. The method showed that quantitative analysis of transport infrastructure effects on biodiversity is quite feasible with already existing techniques, and that holistic impact patterns can be explored by means of relatively simple measures such as literature studies and overlay operations in a GIS. The results (Fig. 6-10) gave us the ability to understand in which types of habitat and for which groups of species transport infrastructure effects were the most negative; information that could help prioritize mitigation and compensatory management activities and inform strategic planning, such as the development of national and regional transport strategies. The method is scalable, and could as well be used at the project level as it is essentially based on assumptions on the habitat quality of nature types and on the ecology and ranging behavior of groups of species. Species that in this study was representing birds and mammals sensitive to habitat loss, fragmentation and disturbance from transport infrastructure.

5. CONCLUSIONS

The importance of EIA and SEA as tools for the development of a sustainable transport infrastructure has increased over the years; as their effectiveness in addressing the environmental dimension of sustainable development has improved substantially. Even so, transport infrastructure effects with detrimental impacts on biodiversity remain elusive due to shortcomings in the concurrent practice of EIA and SEA. Problems with the assessment of biodiversity in EIA and SEA concern primarily impact assessment methods, the treatment of fragmentation and delimitation of the impact assessment study area. From the study it could be concluded that biodiversity impacts were described in text and predicted and evaluated by means of qualitative methods, and this has been so since the introduction of the EIA-directive in 1985. Similarly, the effects of fragmentation were, if considered at all, qualitatively analyzed and described in text. The geographical extent of the impact assessment study area was in EIAs often limited to the construction corridor, which is too small to encompass e.g. fragmentation effects, and the impacts studied are most often the direct impacts resulting from the actual construction, as a timeframe to encompass lagged ecosystem response

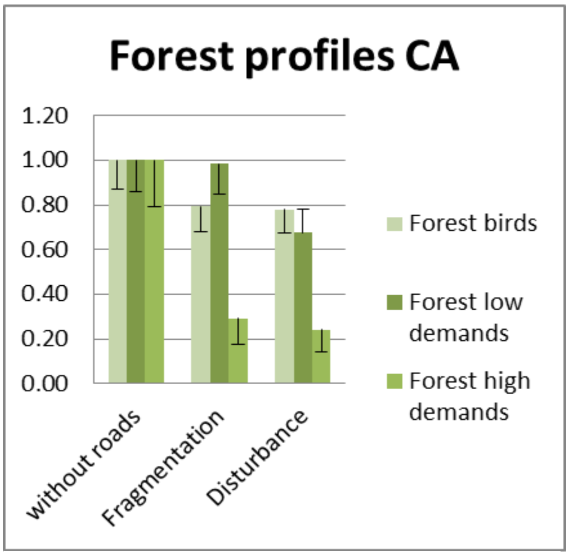

Figure 7 Relative change in total habitat area as a response to fragmentation and disturbance effects

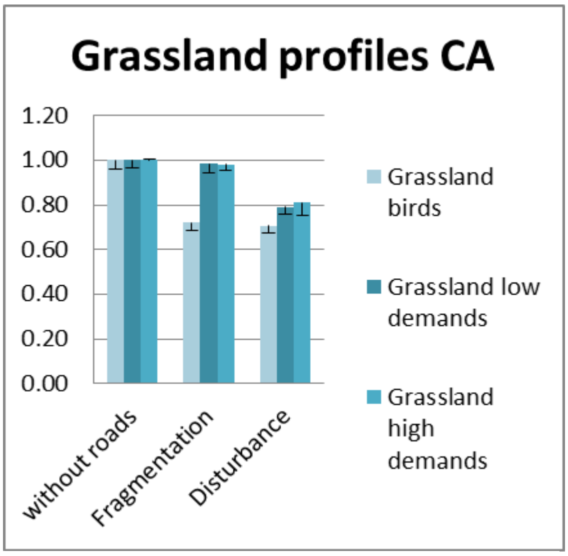

Figure 8 Relative change in total habitat area as a response to fragmentation and disturbance effects

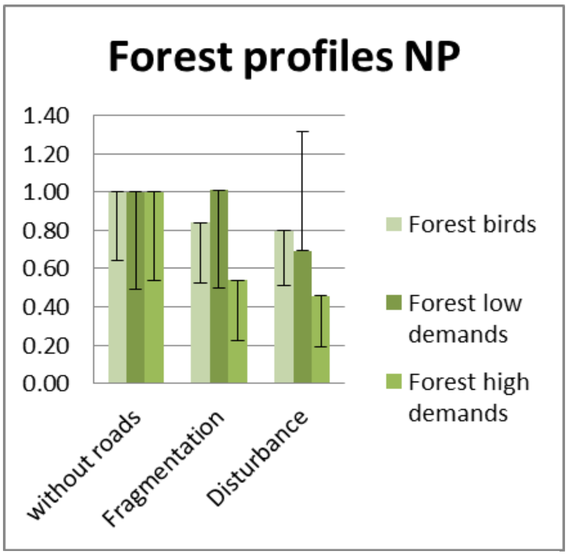

Figure 9 Relative change in habitat sub-division as a response to fragmentation and disturbance effects

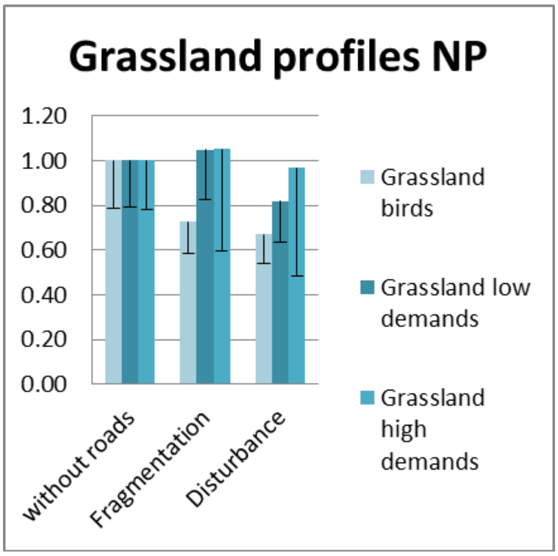

Figure 10 Relative change in habitat sub-division as a response to fragmentation and disturbance effects

seldom is specified. However, a holistic assessment of fragmentation and general transport infrastructure effects (expressed as mean species abundance as a function of distance to roads) was carried out in Paper II using geo-spatial datasets from official sources, literature studies, overlay operations in a GIS and a free categorical map analysis program. This method generated quantitative predictions on the spatial distribution of fragmentation and other transport infrastructure effects in Sweden, information that could complement the qualitative analyses and consultations performed under concurrent practice. Conclusively, in EIA and SEA of transport infrastructure, impact assessment on biodiversity has not yet benefited from quantitative methods despite the availability of tools and techniques. Increased quantitative measurements of how biodiversity respond to transport infrastructure activities would add to the quality of the EIS/ER by providing methods to interpret and describe ecosystem dynamics (e.g. flows in a habitat network), a framework for interpretation and accumulation of data, and by making biodiversity impacts more comprehensible as data can be normalized and presented as relative changes.

6. FUTURE RESEARCH

EIA and SEA would benefit from an increased use of quantitative methods, which would be expected to lead to less uncertainty and a better informational basis on which to take decisions for physical planning. This research has demonstrated how a simple quantitative analysis based on already existing data and techniques revealed that certain nature types and species are most likely more exposed to transport infrastructure impacts than others. This has great potential value; information that can be used to strengthen biodiversity considerations in the development of future transport strategies. A pronounced impact of transport infrastructure constructions is the barrier effect which might have detrimental effects on the viability of populations in in the long term (van der Ree et al. 2005; Shepard et al. 2008; Fu et al. 2010). The barrier effect can be mitigated through the construction of under and overpasses, but such constructions are expensive and their effectiveness relatively uncertain (Seiler et al. 2003; van der Ree et al. 2007; Corlatti et al. 2009). Information on how animals move in the landscape, thus the predicted significance of a barrier is desirable, and likely migration corridors for animals can be modeled using spatial multicriteria (SMCA) techniques (Driezen et al. 2007). SMCA can also be used to design road corridor alternatives for environmental assessment purposes (Geneletti 2003; 2004; Scolozzi and Geneletti 2012 among others), and a case study on suitable road locations based on geological and ecological criteria is intended. Additionally, transport infrastructure is by several authors concluded to be one of several drivers for land use change (Freitas et al. 2010), which has been pointed out as the primary driver of biodiversity loss globally (Rockström et al. 2009). Studies on transport infrastructure as a biophysical predictor for land use change are relatively scarce, but can be approached by classical hypothesis testing through e.g. binary or multinomial regression. Furthermore, literature studies as well as species data can in combination with GIS-based ecological models be applied to model several other transport infrastructure-ecology interactions and to formulate and test hypotheses on transport infrastructure-ecology interactions.

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