

Road ecology in environmental impact assessment



Mårten Karlson ^{*}, Ulla Mörtberg, Berit Balfors

Environmental Management and Assessment Research Group, Department of Land and Water Resources Engineering, KTH Royal Institute of Technology, Brinellvägen 28, 100 44 Stockholm, Sweden

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ABSTRACT

Transport infrastructure has a wide array of effects on terrestrial and aquatic ecosystems, and road and railway networks are increasingly being associated with a loss of biodiversity worldwide. Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA) are two legal frameworks that concern physical planning, with the potential to identify, predict, mitigate and/or compensate transport infrastructure effects with negative impacts on biodiversity. The aim of this study was to review the treatment of ecological impacts in environmental assessment of transport infrastructure plans and projects. A literature review on the topic of EIA, SEA, biodiversity and transport infrastructure was conducted, and 17 problem categories on the treatment of biodiversity were formulated by means of a content analysis. A review of environmental impact statements and environmental reports (EIS/ER) produced between 2005 and 2013 in Sweden and the UK was then conducted using the list of problems as a checklist. The results show that the treatment of ecological impacts has improved substantially over the years, but that some impacts remain problematic; the treatment of fragmentation, the absence of quantitative analysis and that the impact assessment study area was in general delimited without consideration for the scales of ecological processes. Actions to improve the treatment of ecological impacts could include improved guidelines for spatial and temporal delimitation, and the establishment of a quantitative framework including tools, methods and threshold values. Additionally, capacity building and further method development of EIA and SEA friendly spatial ecological models can aid in clarifying the costs as well as the benefits in development/biodiversity tradeoffs.

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Introduction

An effective, smoothly operating transportation infrastructure is a prioritized concern for society. The necessity of functional transportation justifies its consumption of financial resources, and advocates tolerance with its negative concomitants. From an ecological perspective transportation infrastructure induces changes, not only causing local habitat loss, but also disturbances and changes in habitat amount, quality and connectivity and alteration of hydrological processes in the surrounding landscape. 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 thus on biodiversity. Such changes might be beneficial for some species and negligible or negative for others (Fahrig and Rytwinski, 2009; Forman et al., 2003; Hovd and Skogen, 2005; Saarinen et al., 2005), but from many perspectives, transport infrastructure has been found to have detrimental impacts on terrestrial and aquatic ecosystems (Coffin, 2007; Fahrig and Rytwinski, 2009; Forman et al., 2003; Trombulak and Frissell,

2000) and the negative effects of roads on wildlife have even been recognized as one of the major contributors to the global biodiversity crisis (Eigenbrod et al., 2009).

The need for biodiversity considerations in planning and decision making have been stressed in legal frameworks, e.g. the EU Directive 85/337/EEC concerning projects (the EIA Directive), and the EU Directive 2001/42/EC concerning plans and programs (the SEA Directive), as well as in several policy documents, such as the Convention on Biological Diversity (CBD, Article 14). Relevant to the treatment of ecological impacts is also the European Landscape Convention (2004), because it emphasizes the importance of areas without formal protection (Art. 2). The importance of addressing the spatial scale of ecological impacts in planning has been particularly emphasized by the Secretariat of the CBD (2010) and also in the EU Biodiversity Strategy (EC 2011), where preventing habitat loss and increasing functional connectivity between ecosystems across landscapes are seen as important for strengthening biodiversity of protected habitats, for mitigation and adaptation to climate change and to sustain ecosystem services (ES).

The potential utility of adopting an ES-analysis framework for environmental assessment has received much attention from researchers and practitioners recently, and several papers exploring the pros and cons of the ES-concept were published during 2013 (e.g., Baker et al., 2013; Honrado et al., 2013; Karjalainen et al., 2013). Early conclusions

^{*} Corresponding author. Tel.: +46 8 7906558.

E-mail addresses: mkarlso@kth.se (M. Karlson), mortberg@kth.se (U. Mörtberg), balfors@kth.se (B. Balfors).

suggested that an ES approach could facilitate identification of objectives and integration of environmental aspects, but also point out weaknesses; an inability to capture details such as the role and importance of specific species, and highlights a need for even more expert knowledge (Karjalainen et al., 2013). Applying an ES-framework for environmental assessment of transport infrastructure could potentially make their impacts more comprehensible and easier to grasp. Yet, the role of ES in EIA and SEA of transport infrastructure is likely to be more adequate in larger projects and for SEA processes.

The EIA and the SEA directives address direct as well as indirect and cumulative effects of an impact. Assessment of these effects largely depend on which spatial and temporal scales are used in the impact assessment study, and the importance of scales has been pointed out as a challenge to ecological assessment within physical planning (Borgström et al., 2006) and EIA (Gontier, 2007; João, 2002). The choice of scale is highly relevant in ecological impact assessment, as changes induced in one spatial and temporal scale might trigger changes in ecological processes occurring at other scales, with time-lags and nonlinear ecosystem responses (Bastian et al., 2002; Holling, 1973, 1992; Holling et al., 2002; Peters et al., 2004; Peterson et al., 1998). This means that for larger transport infrastructure projects, for example a new railway or highway construction, impact assessment often requires landscape and regional perspectives, as well as long-term temporal perspectives, in order to take into account ecological processes that operate across space and time.

For addressing the spatial scale of ecological impacts in planning, several spatial modeling tools have been developed that have the potential to be useful for biodiversity assessment in EIA and SEA. Such spatial models address ecological processes on different levels; focusing on, for example, ecosystems distribution, fragmentation and rarity (e.g. Cushman et al., 2008; Geneletti, 2003, 2004; Jaeger et al., 2005, 2007); species and community distribution, diversity, abundance, rarity, habitat suitability and connectivity (e.g. Bacaro et al., 2012; Chiarucci et al., 2011; Drielsma and Ferrier, 2009; Gontier et al., 2010; Leathwick et al., 2010; Mörtberg et al., 2007); or both (e.g. Alkemade et al., 2009; Biglin and Dupigny-Giroux, 2006; Di Minin and Moilanen, 2012; Gontier et al., 2006). Another approach is to model prioritized groups of species characterized by their ecological profiles (e.g. Mörtberg et al., 2012; Vos et al., 2001; Zetterberg et al., 2010). The spatial ecological models can be further used as sub-models to integrated assessment tools, where multiple criteria can be considered simultaneously, see for example Li et al. (1999), Geneletti (2006), Coutinho-Rodrigues et al. (2011), Segan et al. (2011), Scolozzi and Geneletti (2012), Bottero et al. (2013). Such spatial modeling tools can be useful in biodiversity assessment in many ways, for instance, for identifying suitable scales for environmental assessment; for localizing, quantifying and visualizing scenarios, conflict areas and suitable areas for mitigation measures; for evaluating planning alternatives, and to increase awareness of cross-scale interactions as well as assessing indirect and cumulative impacts (e.g. González del Campo, 2012). The possibility to address spatial considerations in environmental assessment has also improved substantially with the implementation of the EU Directive 2007/2/EC on Infrastructure for Spatial Information in the European Community (INSPIRE), which involves establishing an infrastructure for spatial information in Europe to support community environmental policies, and policies or activities which may have an impact on the environment. However, Gontier et al. (2006) and Geneletti (2006) found problems concerning the handling of biodiversity in impact assessment. These related to spatial and temporal scales not being duly taken into account and methodological problems, with limited or no use of available spatial modeling tools or other quantitative methods. Further, in a study on the application and effectiveness of the EIA Directive, several problems were identified concerning biodiversity in EIA, including limited consideration of cumulative effects on biodiversity and limited consideration of biodiversity outside Natura 2000 sites (EC 2009). As stated in this study, the lack of quality standards in such situations may render

the application of EIA procedures insufficient in protecting biodiversity. Additionally, a recent communication from the European Commission (EC 2012), states that part of the motivation for the revision of EU law on EIA lies in insufficient quality and analysis of assessments, often resulting in a lack of, or poor quality, EIA data and analysis concerning, *among other topics*, biodiversity. Since the ability to make valid assessments on the environmental impact of a project depends to a large extent on the quality of the information used in the EIA report and the quality of the EIA process, this was considered to lead to ill informed decisions.

Aim

The overall aim of this paper was to analyze and discuss how the treatment of ecological impacts in EIA and SEA has developed since the early enforcement of the EIA directive, specifically focusing on the treatment of transport infrastructure impacts emphasized in the literature of road ecology and landscape ecology. The specific aims of this study were (i) to summarize and structure the scientific literature on ecological impacts of transport infrastructure according to environmental assessment terminology and provide an overview of the Action–Effect–Impact relationships, (ii) to clarify how the treatment of ecological impacts in EIA and SEA of transport infrastructure plans and projects have developed since the introduction of the EIA directive in 1985, and (iii) in view of these findings, discuss future prospects for the treatment of ecological impacts in environmental assessment of transport infrastructure.

Methods

Review of ecological impacts of transport infrastructure

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” describes the attributes of transport infrastructure that induce change in the surrounding environment. “Effect” is 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” is the resulting consequences of those changes.

Content analysis

In order to identify trends in the handling of ecological impacts in EIA and SEA, a second literature review was carried out, this time focusing on articles reviewing the treatment of ecological impacts in Environmental Impact Statements and Environmental Reports (further referred to as EIS/ERs in this study). The review was limited to review articles scrutinizing the treatment of ecological impacts in EIS/ERs of road and railway activities; 8 different articles were selected for this purpose. As the selected review studies were carried out with different aims and methodologies, a content analysis (Kvale and Brinkman, 2009) was employed to circumvent discrepancies. Content analysis is extensively used in social science, and allows for the content of a communication, e.g. a text or an interview transcription, to be systematically analyzed and quantified. The meaning of a text, for example, is coded into categories which make it possible to quantify how often a specific

theme is addressed and compare that frequency to how often other themes are addressed. Content analysis was employed in this study to structure the remarks on the treatment of ecological impacts in the 8 selected articles. However, remarks in the different articles were not expressed in linguistically identical sentences even though one and the same problem was described. Remarks were therefore grouped by value laden similarity into mutually excluding problem categories. The year of publication of each article was plotted along the x-axis in a chart, and the categories of remarks were plotted on the y-axis. Additionally, the timespan in years between the year of publication of the earliest and the most recent article identifying one and the same problem with the treatment of ecological impacts, as described by an EIS or an ER, were also plotted on the x-axis. The problem categories were then further grouped into main problem categories. As such, a pattern emerged during the content analysis which provided additional clarity to the analysis.

Review of recent EISs and ERs

Altogether 7 Environmental Reports and 16 Environmental Impact Statements on road or railway projects, plans or programs produced between 2005 and 2013 in Sweden and the UK were reviewed (Table 1). We chose reports and Impact statements from 2005 to minimize the risk of reviewing EIS/ERs already reviewed by other authors, and because no review articles on this subject from 2006 and onwards were identified at the time of this analysis. The countries Sweden, Scotland, Wales and England were selected because they are all subjected to the EIA and SEA Directives, and because these reports were written in English or Swedish. The categories constructed by content analysis in the review of scientific literature, described in Section Content analysis, were then used as a checklist in the review of recent EIS/ERs. The EIS/ERs were scrutinized, and if a problem as described by the checklist was identified, a cell in the “Problem” column of that category was marked in a chart. For example, if the references to the ecological data used in the assessment were not provided, the intersection of the row “Provisioning of necessary ecological data” and the column “Problem” was marked in a vertical bar. The quantities of vertical bars in the “Problem” and “Not a Problem” columns were then plotted on the x-axis and the categories were plotted on the y-axis. Most of the remarks in the checklist could be identified in the EIS/ERs as either present or absent, as for the above mentioned example. Checklist problems that could not be identified in the EIS/ERs based on presence/absence were

assessed based on how well they were analyzed, relative to the significance of the problem as described in the transport infrastructure–ecology literature. For example, if fragmentation as an impact of a road project was described, but no prediction of where, when and to which extent it might affect the ecological processes in the area, the intersection of the row “Descriptive rather than analytical and predictive assessments” and the column “Problem” was checked in the vertical bar. Additionally, the intersection of “Consideration of habitat fragmentation” and “Problem” was also checked, as fragmentation effects have been recognized to have negative effects on biodiversity and because fragmentation effects can be analyzed by means of several different approaches. The literature reviewed of ecological impacts of transport infrastructure was consulted on such occasions. In several of the environmental assessment reports, methodologies were not fully described but instead references were made to other documents. Whenever possible, these references were identified and reviewed in order to comprehend the quality of assessments, evaluations and field studies undertaken in the different EIA and SEA processes.

Results

Ecological impacts of transport infrastructure

Table 2 presents the main areas of interaction between transport infrastructure and ecological processes, structured using environmental assessment terminology. The “Action” column shows the three main attributes of transport infrastructure that produce effects: its construction, its existence as a physical structure in the landscape and its utilization and maintenance. The “Effect” column lists the general effects relative to the “Action” column, and the “Impact” column describes in text the state of the art knowledge of the consequences resulting from the effects.

The treatment of ecological impacts in environmental assessment

Literature review of the treatment of ecological impacts in transport infrastructure EIS/ERs

The literature review revealed that the treatment of ecological impacts in EIA and SEA have been criticized from mainly three perspectives, presented in Fig. 1 as the main problem categories; 1) EIS/ER contents, 2) ecological knowledge, and 3) methods. The “frequency” bars show the number of articles in which one or several of the problem

Table 1
No. of EIS/ERs reviewed (23 altogether) with information on plan/project type, date and country in which the EIS/ER was carried out.

Country	Type	Activity	ID	Year
Sweden	EIS	New railway	Botniabanan, Bollstabruk–Normyland	2006
		New railway	Ådalsbanan, Härnösand–Veda	2005
		New road	Road 293	2010
		New road	Myre–Stockvik (E4)	2009
		New road	Stockvik–Skönberg (E4)	2009
		New road	Inre kustvägen, Hemmeslöv–Skummeslöv	2012
		Improvement and new local road	Väg 276, Rosenkälla	2011
		Improvement/dualling	E20 Hova	2012
		New road	Sidensjö–Överhörnsås	2012
		New road	Kongberg–Gnarp (E4)	2007
		New road	A90	2007
		New road (bypass)	A82	2009
		New road (realignment)	A737 The Den–Dalry	2013
		Bypass	A737/A738 Dalry Bypass	2013
Scotland	EIS	Dualling program	A9	2013
		New road (link)	A5–M1	2009
		Major improvement	A11	2008
		Bypass	A21 Tonbridge to Pembury	2009
England	Environmental Report EIS	Improvement/widening	A453	2009
		Regional transport plan	South East Wales	2009
		Local transport plan	Wales–Torbay	2006
		National Transport plan	Wales	2009
Wales	Environmental Report	National transport strategy	Wales	2009

Table 2

Overview of transport infrastructure impacts on ecological processes and biodiversity, structured by action–effect–impact relationships according the definition in Catlow and Thirlwall (1976), assuming no mitigation measures are taken. Based on (Ament et al., 2008; Coffin, 2007; Davenport and Davenport 2006; Findlay and Bourdages, 2000; Forman, 2000; Forman and Deblinger, 2000; Forman and Lauren, 1998; Forman et al., 2003; Reijnen and Foppen 2006; Reijnen et al., 1995, 1996; Spellerberg, 1998; Ritters and Wickham 2003; Summers et al., 2011; Trombulak and Frissell, 2000).

Action	Effect	Impact
Construction	Habitat loss	Habitat is converted into road/railway construction. Local impacts on populations, potential impacts on meta-populations. Increased landscape subdivision, especially from linear constructions, resulting in reduced connectivity and increased isolation. High risk for long term biodiversity loss.
	New habitat	Road and railway structures constitute attractive habitat for many species. Favorable for generalist and disturbance tolerant species, unfavorable for species with specific habitat requirements, with certain exceptions especially concerning plants and invertebrates. Road and railway habitat also function as conduits for dispersal. Highly uncertain consequences.
Physical structure in the landscape	Linear landscape elements	Alteration of habitat size, increased habitat fragmentation and changes in connectivity lead to species specific responses; avoidance or adaption with uncertain consequences; enhanced or reduced colonization rates and movement capacity in terrestrial and aquatic ecosystems. If perceived as a barrier, isolation of populations increases and have overall negative consequences on biodiversity.
	Change of hydrological patterns	Alteration of hydrological processes: runoff, recharge, erosion and sedimentation. Alteration of microclimatic conditions. Impacts may accumulate over time with increased risk for long term deterioration of aquatic habitat and biodiversity loss.
	Land use change	Transportation networks provide socioeconomic incentives for different kinds of land use. Indirectly a driver for habitat loss and fragmentation thus increasing risk for biodiversity loss.
Utilization and management	Road mortality	Reduced population densities at local scales, uncertain consequences at meta-population scales. Evidence for road mortality as population sinks for some species.
	Introduction of exotic species	Linear elements often function as conduits for dispersal. Vehicles can function as vectors for exotic species, significantly increasing colonization ranges and rates. Some species have been categorized as invasive and sometimes threats to native species, but consequences are overall highly uncertain.
	Disturbance	Altered noise levels, light pollution, visual disturbance, increased human access and activities. Reduced species richness and abundance in road adjacent habitat. Impacts on many biodiversity components even though negligible or positive consequences for some species. Choice of management regimes crucial to certain biodiversity components.
	Chemical contamination	Local contamination of heavy metals and chemicals to air, water and soil. Pollutants may accumulate within biomass and remain within terrestrial and aquatic ecosystems. Regional precipitation of dust and airborne pollutants with impacts on recipient ecosystems and on human health. Pollutants may accumulate within biomass and remain within ecosystems. Uncertain consequences.

categories were noted. The “timespan” bars show how long a “Problem” had been considered problematic, based on the number of years that passed between the first and the last time it was mentioned in a research article. 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. Of the problems related to the EIS/ER contents, Fig. 1 shows that the largest share of remarks (the frequency bars in Fig. 1) were attributed to the following problems: low commitment to monitoring (problem 1f, five articles), the description of methods used and the assessment of impact significance (problems 1c and 1d, three articles respectively), the consideration of impacts on valuable non-protected areas (problem 2c, five articles), the consideration of indirect and cumulative impacts (problem 2f, four articles), the treatment of habitat fragmentation (problem 2g, four articles), assessments being descriptive rather than analytical and predictive (problem 3c, four articles) and the use of quantitative methods (problem 3d, five articles). Similarly, the persistence of problems with the treatment of ecological impacts were measured as the number of years that had passed between the first and the last time a problem was mentioned in a research article (the timespan bars in Fig. 1), were the same as those that received the largest share of remarks: problem 1f (14 years), problems 1c, 1d, 2c and 3d (10 years respectively) and problems 2f, 2g and 3c (nine years respectively).

Review of recent EIS/ERs

The number of EIS/ERs in which a problem was identified are given by the red bars, and the number of EIS/ERs in which a problem was not identified are given by the blue bars (Fig. 2). Three of the 17 problems 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 persisting

problems was, however, not equal between the main problem categories 1) EIS/ER contents, 2) ecological knowledge and 3) methods. Regarding the EIS/ER content problems (main problem category 1), Fig. 2 shows that none of the problems identified in the review of scientific literature could be seen as highly problematic for ecological considerations in EIA and SEA at the time of this EIS/ER review (2013), as they were reported in less than half to none of the reviewed reports. Of the problems related to ecological knowledge (main problem category 2), Fig. 2 shows that the three most criticized problem categories identified in the review of scientific literature; consideration of valuable non-protected areas, the treatment of indirect and cumulative impacts and the treatment of habitat fragmentation was a problem in three, 17%, 48% and 43%, of the reviewed EIS/ERs respectively. Fig. 2 also shows that provisioning of necessary ecological data was a problem in 52% of the EIS/ERs, primarily those from Sweden. This critique, first given in Treweek (1996), thus still applied to the Swedish EIS/ERs, but not to the EIS/ERs from the UK, in which field study reports and other information that concerned the ecological data used in the EIA/SEA study were referenced in a way that made them easily accessible. Fig. 2 further shows that problems with the delimitation of the impact assessment study area were identified in 68% of the EIS/ERs, that consideration was made for red-listed species in 100% of the EIS/ERs, and that the size of the project was specified in 87% of the EIS/ERs. Regarding problems related to methods used for impact assessment (main problem category 3), Fig. 2 shows that assessments being descriptive rather than analytical and predictive (problem 3c) was identified as “Problem” in 48% of the reviewed EIS/ERs, and that the scarce use of quantitative methods (problem 3d) was identified as “Problem” in 87% of the reviewed EIS/ERs. Further, the quality of field surveys (problem 3b) was identified as “Not a problem” in 100% of the EIS/ERs that presented a field survey methodology that could be scrutinized, and information about the ecological conditions in the project area (problem 3a) was presented in all of the reviewed EIS/ERs.

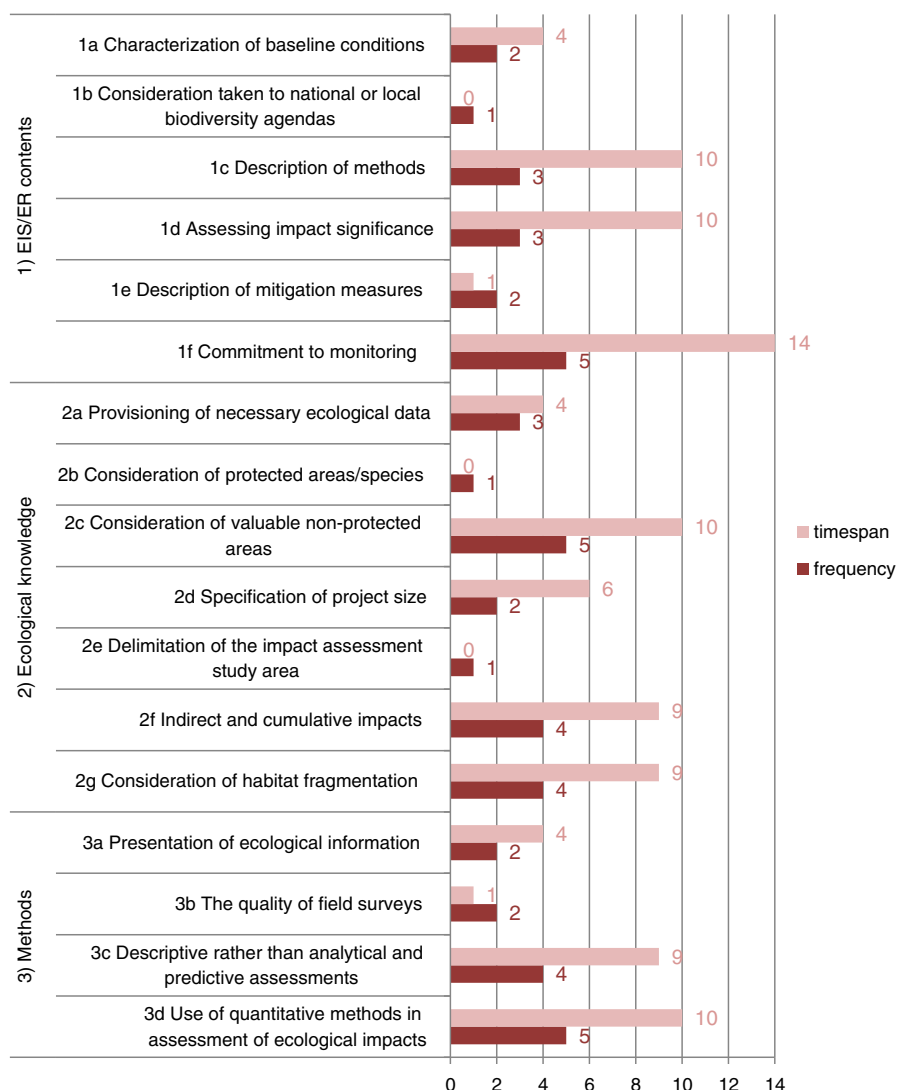


Fig. 1. Problems with the treatment of ecological aspects in EIS/ERs that were identified in the scientific literature. The frequency-bars show the number of articles in which a problem has been identified, and the timespan-bars show the number of years between the earliest and the most recent article in which one and the same problem was identified. Based on Treweek (1996); Thompson et al. (1997); Atkinson et al. (2000); Byron et al. (2000); Cooper and Sheate (2002); Gontier et al. (2006); Geneletti (2006); Lundberg et al. (2010).

Discussion

Ecological impacts of transport infrastructure

The “Action–Effect–Impact” relationships in Table 2 are a broad overview of how transport infrastructure interacts and induces changes in ecological processes and how it impacts on biodiversity. The Action–Effect–Impact overview in Table 2 depicts the main impact-patterns described in the literature and the relationships between impacts and the different stages of a projects life cycle, which for EIA and SEA purposes would help to understand which ecological impacts to focus on, and when, during the planning process those impacts would be necessary to address. It is clear from Table 2 that only a minor share of the known impacts generated by transport infrastructure is related to the actual construction, but that most impacts are indirect and result from the construction and design of the physical structure, and its utilization and management.

The construction of transport infrastructure

Concerning the effects of habitat loss and the creation of new habitat during construction of transport infrastructure, the loss of habitat is,

from a biodiversity perspective, inevitably negative because habitat amount is positively correlated to core ecological processes and to indicators like species richness and abundance (for an overview, see e.g. Hanski and Gilpin (1991)). Several authors have put forward the effects habitat loss and fragmentation as main causes for overall negative effects on wildlife (Frair et al., 2008; Jaeger et al., 2007). Fragmentation effects caused by transport infrastructure are considered to be increased by barrier effects, and in combination they could substantially reduce the connectivity of habitat networks. This can lead to reduced colonization rates both for terrestrial and aquatic species (Rico et al., 2007; Shepard et al., 2008; van der Ree et al., 2007, 2009). At the same time, the creation of new habitat provides an opportunity to enhance and strengthen biodiversity in the landscape (Lennartsson and Gylje, 2009; Ottosson et al., 2012; Runesson, 2012). Transport infrastructure environments can be highly similar to biotopes such as natural grasslands and sandy and gravelly environments, increasing these habitat types. Several rare and specialist plant species that would otherwise be found in habitat types of increasing rarity, have found refuge in road side verges and switchyards. The high plant species richness may also attract pollinators, and a high invertebrate diversity is also common (Hovd and Skogen, 2005; Lennartsson and Gylje, 2009). Another effect

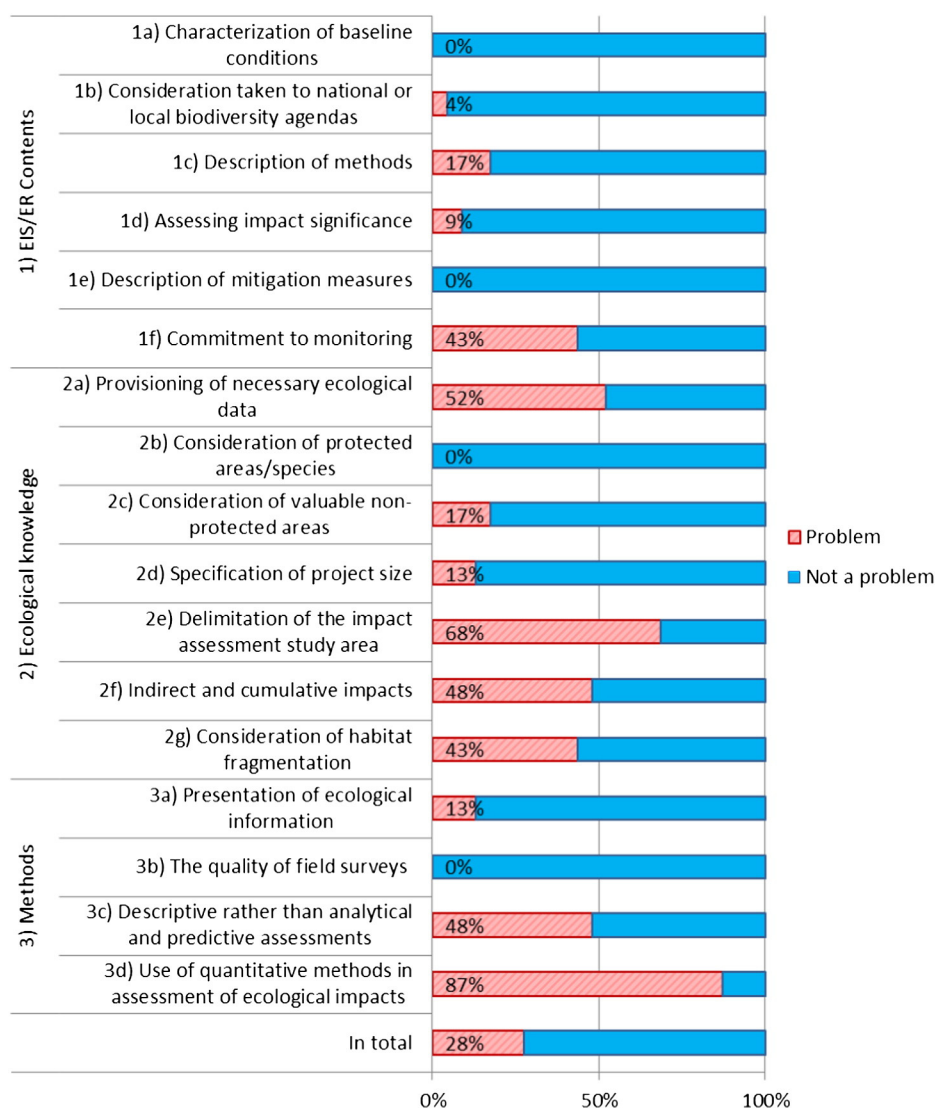


Fig. 2. Results from review of the treatment of ecological impacts in EIS/ERs produced between 2005 and 2013 in Sweden, Scotland, Wales and England. Of the documents reviewed, 7 were Environmental Reports and 16 were EISs, of which 17 concerned road projects, 2 concerned railway projects and 4 plans concerned both modes of transportation. This review was carried out using the problems identified in the review of scientific literature (Fig. 1) as a checklist.

of transport infrastructure habitat that has been stressed in the literature is how it can function as a dispersal conduit for some species. Where non-native species have been able to spread along transport infrastructure corridors, it has in some cases had negative consequences on species richness and abundance (Forman et al., 2003; Gelbard and Belnap, 2003; Hulme, 2009; Spellerberg, 1998; Stohlgren and Schnase, 2006; Trombulak and Frissell, 2000; Yan et al., 2001).

The physical structure of transport infrastructure

Roads or railways are a barrier to movement, to a greater or lesser extent, for most species without the ability to fly, but even to some flying animals as well. 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 (Corlatti et al., 2009; Diaz-Varela et al., 2011; Mata et al., 2009; van der Ree et al., 2007, 2009), even though their effectiveness for biodiversity remains uncertain. Likewise, well designed drainage culverts could most likely serve as underpasses for migrating fish, amphibians and small mammals (Clevenger et al., 2001; Feurich et al., 2012; Glista et al., 2009). Furthermore, transport infrastructure as a novel physical

structure in the landscape has effects on hydrology. Rainwater run-off and infiltration patterns are altered causing increased loads of sedimentation, nutrients and contaminants, which can have negative impacts on downstream biota. Transport infrastructure has also been proposed to indirectly drive 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 (Balfors et al. 2005; Freitas et al., 2010; Fu et al., 2010; Olsson, 2009).

The use and management of transport infrastructure

Several studies conclude that species richness and abundance are reduced within certain distances from roads (Ament et al., 2008; Biglin and Dupigny-Giroux, 2006; Bissonette and Rosa, 2009; Boarman and Sazaki, 2006; Coffin, 2007; Eigenbrod et al., 2008, 2009; Fahrig and Rytwinski, 2009; Forman and Deblinger, 2000; Forman and Lauren, 1998; Forman et al., 2002; Helldin and Seiler, 2003; Huijser and Bergers, 2000; Jackson and Fahrig, 2011; Jaeger et al., 2005; Olsson, 2009). There is however a great variation in species responses to transport infrastructure, and a few contrasting views on what could be causing the density depressing effect of transport infrastructure have been put forward in

the literature. Some studies focus on disturbance regimes since, primarily, noise and vibrations appear to have a distinct negative effect on species richness and abundance of birds, amphibians and reptiles (Eigenbrod et al., 2009; Forman and Deblinger, 2000; Forman et al., 2003; Hoskin and Goosem; Reijnen et al., 1995; Reijnen et al., 1996). Disturbance effects can nevertheless be mitigated by, for example, enforcing lower vehicle speed or installing noise screens in sensitive areas. Furthermore, road mortality is addressed by some researchers as the most significant cause for density depression (Jackson and Fahrig, 2011; Summers et al., 2011) and species life history traits are to a large extent considered to explain why some species are more vulnerable to road mortality than others (Fahrig and Rytwinski, 2009; Rytwinski and Fahrig, 2012).

Utilization and management of transport infrastructure cause chemical contamination of air and water through vehicle exhaust, use of pesticides and de-icing salts (Forman and Lauren, 1998; Forman et al., 2003; Jones et al., 2000; Trombulak and Frissell, 2000). Management regimes such as mowing intensity and techniques as well as use of pesticides and fertilizers also greatly influence how plant and invertebrate diversity will develop in an area. Mowing regimes are predominantly designed to minimize wildlife accidents and other threats to traveling vehicles rather than to enhance biodiversity or to combat invasive species. Still, management regimes have great potential to enhance viability of prioritized biodiversity components such as red listed species and invertebrate habitat suitable for pollinator species, and thus associated ecosystem services (Ottoosson et al., 2012; Runesson, 2012). However, the quantities of invertebrates killed by traveling vehicles are immense, and questions have been raised over whether constructing attractive invertebrate habitat along roads would increase species richness and abundance or function as population sinks (Swedish Biodiversity Centre, 2012).

In summary, Table 2 highlights the complexity of assessing the environmental impacts of transportation infrastructure. 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, as ecological processes often operate at landscape or regional scales and require long-term outlooks to be adequately assessed. At the same time, the construction, localization and design, as well as management of transport infrastructure provides opportunities to enhance biodiversity values by supporting ecological processes. In this way, much of the negative effects on wildlife might be avoided, mitigated or to some extent compensated.

Ecological impacts in environmental assessment

Since the EIS/ERs were reviewed using a checklist essentially based on quality criteria derived from scientific literature, any “improvement” or “deterioration” of the treatment of ecological impacts should be viewed relative to those criteria. The early reviews of ecological impacts in EISs by Treweek (1996) and Thompson et al. (1997) seemed to have had notable influence on the formulation of problem categories, and therefore the results of this study could partly be seen as a follow-up to those initial studies. The sample consisted of 23 EIS/ERs in total, gathered from Sweden and the UK. Despite being subject to the same legal frameworks for environmental assessment, the structure, content and analytical methods used between the countries varied greatly. The UK countries all followed the same EIA and SEA guidelines, and were similar in structure and content but with differences in quality. The Swedish EIS/ERs were notably different, and structure, content, methods and quality varied even within that group of documents. The EIS/ERs from Sweden were shorter in length and contained less information than the EIS/ERs from the UK. Additionally, different parts of the plan/project area and different aspects of impacts were studied in a step-wise process, resulting in the information of impacts becoming scattered over several different EISs. Further, in the Swedish EIS/ERs important

information such as field study reports and methodologies, or EISs in which impacts had been analyzed previously where generally not available from a single entry, which made it difficult to comprehend the full environmental assessment process as well as to back-trace which decisions had been taken earlier in the road/railway planning process. As referencing in general was poor, documents of interest were not always published online, or belonged to other organizations, institutions or private companies; relevant sources of information were not easily accessible for a few of the Swedish EIS/ERs reviewed.

In summary, the EIS/ERs following British-developed guidelines (The Design Manual for Road and Bridges (DMRB), volume 2) were composed in good compliance with the EU-directives. This might be due to the DMRB guidelines, which advocate very precise criteria for impact assessment and evaluation. The Swedish equivalent does not recommend equally specific evaluation criteria and are, overall, less precise. However, the strict compliance with the DMRB guidelines appeared to not only have streamlined the structure of the EIS/ERs, but also the actual impact assessment study and the resulting findings. In contrast to the Swedish reports, in which the focal impacts varied in each and every EIA or SEA, the UK EIS/ERs consistently brought up the same aspects and often very similar objectives for mitigation. An example of this is that key species (e.g. badgers, bats and otter) and vegetation types (e.g. hedgerows) were screened on a regular basis, and if deemed negatively affected, standardized mitigation measurements were suggested (e.g. plant new hedgerows to reconnect isolated stands).

Problems with EIS/ER contents

Of the 17 problem categories identified in the scientific literature, six concerned the EIS/ER contents (Fig. 1). The result of the EIS/ER review show that they were reported in less than half or none of the EIS/ERs (Fig. 2). This is a strong indication of an improvement since Treweek (1996) article on ecology in Environmental Impact Assessment, where several content-related problems were listed. For instance, the earlier problems concerning characterization of baseline conditions and description of mitigation measures could not be identified in any EIS/ER. Fig. 2 also shows a considerable improvement in the commitment to monitoring activities. In all of the ERs (related to SEA), and in most of the EISs (related to EIA), an environmental monitoring program was presented. Monitoring is not required by the EIA Directive, but is required by the SEA Directive. Hence, with regard to the EIS/ER content related problems listed in Fig. 1, the current results (Fig. 2) indicate a substantial improvement.

Problems with the ecological knowledge in EIS/ERs

Of the problems relating to ecological knowledge (main problem category 2), Fig. 2 shows, when compared to Fig. 1, improvements in the three most criticized problem categories identified in the literature review; that consideration of valuable non-protected areas had improved substantially, whereas the treatment of indirect and cumulative impacts as well as of habitat fragmentation had improved slightly. Furthermore, Fig. 2 shows that problems with the delimitation of the impact assessment study area were identified in 13 reports. The choice of spatial and temporal delimitation strongly influences which impacts will be detected and assessed. As shown in Table 2, several of the listed impacts concerned biodiversity components that are not necessarily located within the project area, like water bodies receiving increased runoff, the effects of fragmentation on meta-population dynamics, the consequences of road mortality or reduced breeding success. The potential to successfully address many of the indirect impacts is highly dependent on the delimitation of the study area and the time frame specified for the impact assessment study (problem 2e). Thus, if this were revised properly, it could generate improvements in the treatment of ecological impacts in EIA and SEA. Concerning the ERs (related to SEA) the scales of the plans themselves were large and therefore the spatial scale may not seem to constitute a problem from an ecological perspective. 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. The impact assessment was concentrated within this corridor, but expanded if deemed necessary or by predefined rules of thumb, for example, within a certain proximity to protected areas or if protected species were observed in initial field surveys.

In the EIS/ERs produced in the UK the study area was expanded outside the road corridor, sometimes generously. However, only in a few reports was this expansion related to the nature of the actual impacts the projects were estimated to cause. Instead, the road corridor was thoroughly investigated through field studies, during which features of high conservational value were discovered, which led to the study area being enlarged and to additional surveys and field studies. However, any analyses or even reflections on the spatial relationships of the attribute and features in the study area and the surrounding landscape were rare. By contrast, in EIS/ERs produced in Sweden the study area was delimited differently from case to case, and there was no apparent use of a framework for delimitation. A common attribute of the Swedish reports was a more holistic description of ecological values, with less emphasis on designated areas and more on the wider landscape. In one EIS the study area was adapted to the hydrological processes in the area, and barrier effects of the proposed motorway were considered together with the pre-existing barriers and the overall connectivity in the area (Sweden, E4). But in most EIS/ERs the study area remained within the road corridor, and for some projects it could be considered to be too small. 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. With regard to ecological knowledge in EIS/ERs, Söderman (2006, 2009) identified the treatment of indirect and cumulative impacts as problematic. The results of this review showed a slight improvement as indirect and cumulative impacts were now considered in more than half of the reviewed reports. The reports produced in the UK contained a separate assessment of cumulative impacts where analyses of these effects were presented by means of a qualitative matrix. In the Swedish reports, cumulative impacts were mentioned in some reports, but not analyzed. Furthermore, fragmentation and connectivity were mentioned in more than half of the EIS/ERs but only qualitatively analyzed in a few, and not quantified in any. In a few reports, fragmentation effects were described, but only in general terms or vaguely related to the project. 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, these can be seen as remaining critical problems even though certain improvements were noticed.

Methods for addressing ecological impacts

Regarding methods used for impact assessment (main problem category 3), Fig. 2, in comparison with Fig. 1, shows that the problems of assessments being descriptive rather than analytical and predictive (problem 3c) and that of the use of quantitative methods (problem 3d) remain problematic. Ecological impacts were qualitatively predicted in all but 1 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. Fig. 2 also shows that no problems with the quality of field surveys could be identified in any of the EIS/ERs. Extensive fieldwork had been carried out in all of the EIA processes. In the reports from the UK up to 12 different surveys (road A21, UK) could be carried out preceding a project, depending on how many prioritized biodiversity components were identified during screening surveys. Data retrieved from field studies are of course invaluable and help to reduce uncertainty in impact prediction. Such data in combination with quantitative methods could also be used to relate ecological processes to a larger context, and thereby reduce uncertainties of predicted effects outside the project corridor, and including long term effects. However, a quantitative prediction was 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 Tweek (1996) as well as by Geneletti (2006) and Gontier et al. (2006) and can still be considered to be problematic.

As indicated by Fig. 2, the most recent EISs (produced in Sweden 2012 and Scotland 2013) employed quantitative techniques for the sake of assessing impacts on ecological processes. Still, 20 out of the 23 reviewed EIS/ERs relied on desktop studies and expert opinions, which can be seen as highly problematic because transport infrastructure may cause impacts which affect ecological processes on wide spatial and temporal scales, and may act as a driver for biodiversity loss (e.g. Pearson 2013). Moreover, even if habitat fragmentation was mentioned in more than half of the EIS/ERs, the effects were poorly analyzed and none did so by means of quantitative techniques. In a few reports project land-take was quantified, but land conversion and land use change were mostly treated in a land-use section, and not considered to have implications for ecological processes and biodiversity. Quantitative methods, such as the spatial ecological models mentioned in the Introduction, have the potential to predict and assess ecological impacts at multiple scales in space and time, and also to help design ecologically well-informed mitigation measures. However, no EIS/ERs used spatial ecological models for assessment of ecological impacts. Similar critique was given by Geneletti (2006), who concluded that the use of GIS was limited to mapping and display. The results of this study supports this; in all but two EIS/ERs, in which overlay operations were used, GIS was only used for mapping and display of various landscape features. Furthermore, integrated assessment tools for multi-criteria decision aids, that can address and integrate quantitative and qualitative issues as well as stakeholder perspectives, were not used. Such tools have, though, been commonly applied in impact assessment contexts in some countries (Janssen, 2001).

In summary, the spatial distribution of impacts together with the lagged and non-linear response to change by many recipients, species or ecosystems (Findlay and Bourdages, 2000; Holling, 1973; Peters et al., 2004; Trombulak and Frissell, 2000), can be seen as a significant challenge to the prediction of ecological impacts, not least for EIA and SEA. In order to improve the treatment of ecological impacts concerning biodiversity and ecosystem services, a way forward may be to standardize the spatial and temporal scales used for impact prediction and assessment, follow-up studies and monitoring in EIA and SEA of transport infrastructure projects. Additionally, capacity building and further method development of EIA and SEA-friendly spatial ecological models would be useful. The use of quantitative methods for spatial appraisal would enable a higher replicability of studies, enhance transparency and improve handling of uncertainties, as well as mainstreaming the production of environmental/ecological data. Robust frameworks for quantitative ecological analysis and interpretation of ecological impacts including thresholds, reference values and standards indicating acceptable levels of impact on ecological processes would also facilitate and mainstream ecological assessment, and help clarify the costs as well as the benefits in development/biodiversity tradeoffs.

Conclusions

Transport infrastructure has direct and indirect impacts on ecological processes, including habitat loss, fragmentation and degradation, increased mortality and the creation of new habitat. The literature review revealed that ecological impacts from transport infrastructure could be related to three distinct phases of the infrastructure lifecycle: its construction, the design of its physical structure in the landscape and its utilization and maintenance. Furthermore, problems with the treatment of ecological impacts in EIA and SEA could be structured into three main problem categories: EIS/ER contents, ecological knowledge, and methods. Of the 17 problem categories identified in the literature review, three were identified in more than half of the current review of recent EIS/ERs. Improvements that had occurred, relative to the list of problems synthesized from the literature, were most evident not only in the problems related to the EIS/ER contents, but also in the problems “Consideration of valuable non-protected areas”, “Presentation of ecological information” and “The quality of field studies”. Remaining problems with the treatment of ecological impacts in EIA and SEA were found in the main problem categories “Ecological knowledge” and “Methods”. More specifically, the poor treatment of habitat fragmentation, that the impact assessment study area was delimited in space and time, without consideration of ecological processes, and that of quantitative methods, though available, were not used. Suggested ways to improve the handling of ecological considerations in EIA and SEA could be to standardize the spatial and temporal scales used in prediction, assessment and monitoring of ecological impacts. Further method development of spatial ecological models relevant to EIA and SEA practice, and robust frameworks for quantitative ecological analysis and interpretation including standards indicating acceptable levels of impact on ecological processes would also facilitate and standardize assessment of ecological impacts.

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Mårten Karlson is a PhD-student at the Department of Land and Water Resources Engineering, at KTH Royal Institute of Technology in Sweden. He is a member of the research group Environmental Management and Assessment, and focuses on environmental assessment of transport infrastructure impacts on biodiversity. Mårten's research is primarily based on landscape ecological theories and the effective application of these in Environmental Impact Assessment and Strategic Environmental Assessment. GIS and spatial data sources constitute the theoretical landscape in which patterns and processes can be explored and analyzed and hypothesis testing can take place.



Ulla Mörtberg is an assistant professor in land and water resources engineering, active in the Environmental Management and Assessment Research Group, KTH Royal Institute of Technology. Her research concerns landscape ecology and environmental systems analysis applied in the development of methods and tools for the integration of scientific knowledge in planning and decision making.



Berit Balfors is a professor in Environmental Assessment Analysis and Head of the Department of Land and Water Resources Engineering, Royal Institute of Technology (KTH). She is in charge of the Division of Environmental Management and Assessment. Her research field encompasses tools for environmental systems analysis in particular EIA and SEA.