Chapter 2. Key Ecological Concepts

This chapter introduces some of the major ecological concepts that aid an understanding of the large-scale effects of infrastructure on wildlife: the concepts of landscape, scale and hierarchical organisation; the process of habitat fragmentation; the importance of habitat connectivity and corridors for animal movement; and metapopulation dynamics. There is a focus on landscape pattern and structure, particularly how these interact to determine the impact of infrastructure on wildlife. The chapter emphasises the importance of planning at a landscape scale and explains why the use of a broader, landscape ecological approach may shed new light on barrier and isolation effects.

Habitat fragmentation caused by transportation infrastructure is an issue of growing concern (Prillevitz, 1997). Possible effects of fragmentation on wildlife have been recognised and an impressive amount of empirical studies illustrate the widespread impact on species and ecosystems (see Chapter 3). The growing demand for information on efficient mitigation has, however, highlighted that the current understanding of the long-term, large-scale ecological consequences of infrastructure provision is insufficient (Treweek *et al.*, 1993; RVV, 1996; Seiler and Eriksson, 1997; Forman, 1998). It is apparent that impacts cannot be evaluated from a local perspective alone. Infrastructure planning must therefore involve a landscape wide, holistic approach that integrates technical, human and ecological requirements. Landscapes and habitats are two fundamental aspects that infrastructure planners must consider. This chapter clarifies the definitions of these, and other important terms and concepts relevant to habitat fragmentation.

2.1. LANDSCAPES AND HABITATS

The definition of the term landscape varies considerably between European countries and scientific domains. For the purposes of this document, it is defined as 'the total spatial entity of the geological, biological and human-made environment that we perceive and in which we live' (Naveh and Lieberman, 1994). Landscapes are composed of a mosaic of individual patches embedded in a matrix (Forman, 1995). The *matrix* comprises the wider ecosystem or dominating landuse type in the mosaic and usually determines the 'character' of the landscape, e.g. agricultural, rural, or forested. Landscape patches are discrete spatial units that differ from each other due to local factors such as soil, relief, or vegetation e.g. an area of forest surrounded by grassland, or a pond within a forest. Landscape patches may also be termed 'habitat'. In ecology, the term *habitat* is a species-specific concept of the environment in which a plant or animal finds all necessary resources for survival and reproduction (Whittaker et al., 1973; Schaefer and Tischler, 1983). The size of a habitat is therefore entirely dependant upon the individual species' requirements: it can be anything from a pond, a meadow, a forest or even the entire landscape mosaic. The diversity of habitats within a landscape and the spatial arrangement of individual habitat patches together determine the biodiversity value of the landscape (Gaston, 1998). Biodiversity denotes the total variation among living organisms in their habitats, including the processes that link species and habitats.

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2.2. LANDSCAPE CHANGE AND HABITAT FRAGMENTATION

Historically, human activities (driven by politics, economics, and cultural traditions) have altered landscape patterns, habitat quality and the 'natural' distribution of species (Stanners and Bourdeau, 1995; Jongman *et al.*, 1998). Across Europe, traditional small-scale landuse has been replaced by intensified methods that require large, homogeneous production units (Burel, 1992; Jedicke, 1994; Ihse, 1995; Skånes and Bunce, 1997). In modern rural landscapes, wildlife habitats have been reduced to small remnants scattered throughout the intensively used matrix. In addition, extensive natural areas, *e.g.* open marshland or contiguous forests, have been increasingly fragmented by infrastructure including roads, railways, waterways, drainage ditches, and power lines (*e.g.* Bernes and Grundsten, 1992; Kouki and Löfman, 1999; and Figure 2.1). As a result, species have come to depend on increasingly smaller patches of remnant semi-natural habitat and green corridors such as hedgerows, wooded field margins, infrastructure verges and small forest patches.





Forest roads in Northern Sweden



Wooded road verges and hedgerows in northern Germany

Figure 2.1 - Landscape change due to fragmentation and loss of connectivity. Top -Increase in forest road network in the Jokkmokk area in northern Sweden between 1935 and 1988 (after Bernes and Grundsten, 1992). Lower - Loss of vegetated corridors (tree rows, hedgerows, road verges) in the agricultural landscape of northern Germany between 1877 and 1979. (After Knauer, 1980) Together, forestry, agriculture and urbanisation have significantly reduced landscape heterogeneity and the extent of 'natural' habitats (Richards, 1990; Jongman, 1995; and Figure 2.2). Globally, this loss of landscape heterogeneity and the fragmentation of large, previously undisturbed habitats has created a major threat to biodiversity (Burgess and Sharpe, 1981; Wilcox and Murphy, 1985; Gaston, 1998). To promote the sustainable use of landscapes, people must learn to think and plan at a larger scale, integrating the local considerations into a broader functional context (Forman, 1995; Angelstam, 1997).



Figure 2.2 - Four types of landscapes that differ in the degree of human impact: A) A natural forested landscape containing a variety of natural ecosystems and habitats with little or no human influence; B) A mosaic, rural landscape where pastures, fields blend with forests that connect through hedgerows and strips of woody vegetation along small watercourses; C) A landscape dominated by agriculture and extensive land cultivation where remnants of the natural vegetation may be found in gardens and along infrastructure verges; 4) An urban landscape, strongly affected by infrastructure and built-up areas with little or no space for wildlife. (Drawings by Lars Jäderberg)

Habitat fragmentation is a process that splits contiguous habitat into smaller patches that become more and more isolated from each other. At the beginning of the fragmentation process, the loss of habitat is the driving force reducing species diversity in the landscape. Towards the end of the process, isolation effects become more important (Harris, 1984). Empirical studies indicate that the number of species drops significantly when more than 80% of the original habitat is lost and as habitat remnants become isolated (Andrén, 1994). The exact fragmentation thresholds depend on species' habitat requirements and mobility, and the mosaic pattern of habitats in the landscape. Where habitat remnants are connected through 'green' corridors or by small, suitable patches which serve as stepping stones (see Section 2.5), isolation effects may be minimised. The landscape may then support a higher diversity of species than would be expected from the overall area of remnant habitat. However, where roads or railways cause additional separation of habitats (see Chapter 3), critical thresholds of fragmentation may be reached much earlier (Figure 2.3). It is essential that infrastructure planning should therefore consider the existing degree of fragmentation in the landscape, species' characteristics and the ecological scale at which the fragmentation effect may be most severe (Seiler and Eriksson, 1997).



Figure 2.3 - (1) Fragmentation of an animals' habitat (shaded areas) reduces the ability of individuals to move across the landscape. (2) Some connectivity may be sustained through small habitat fragments or corridors. (3) Infrastructure imposes additional movement barriers and strengthens the isolation effect caused by habitat fragmentation. (4) Mitigation measures such as fauna passages and integrated road verge management can help to re-establish or even improve habitat connectivity in the landscape.

The consequences of habitat fragmentation to wildlife are complex, as species respond differently to the loss and isolation of their habitat. In general, species with limited mobility, large area requirements, or strong dependence on a certain type of habitat will be among the first to suffer the effects of habitat loss and isolation. These species generally respond to habitat fragmentation by modifying their individual behaviour patterns. Conversely, species that are abundant at a landscape scale, that utilise a variety of habitats and are more resilient to disturbance may not be affected so significantly. Although infrastructure may represent a significant barrier to their movement, local populations can be sustained so long as the habitat remnants remain sufficiently large. Isolation effects manifest themselves in this group of species through long-term demographic and genetic change within the population. Applying this knowledge in infrastructure planning is the key to preventing the ultimate consequence of habitat fragmentation - species extinction. In terms of defragmentation strategies, wide-roaming species will benefit most from improved habitat connectivity whilst for the smaller and less mobile species, more effort should be put into protecting and enlarging local existing habitats (Fahrig and Merriam, 1994).

2.3. METAPOPULATIONS, SINKS AND SOURCES

Two ecological theories, regarding metapopulations (Levins, 1969) and sink and source population dynamics (Pulliam, 1988), contribute to the understanding of the complex processes of colonisation and extinction of populations in the landscape. These approaches help ecologists to predict the wider effects of habitat fragmentation and design effective strategies for the conservation of fragmented populations (Harris, 1984).

A *population* is a group of individuals of the same species that live in the same habitat, and breed with each other. When a habitat is fragmented, a system of local populations is formed. Where these are located close enough to permit successful migration of individuals, but are sufficiently isolated to allow independent local dynamics, the system is called a *metapopulation* (Hanski and Gilpin, 1991). The migration of individuals between the local *source* (where the number of births exceeds the number of deaths) and *sink* (with a negative birth to death ratio) populations has a stabilising effect on metapopulation dynamics (Pulliam, 1988). However, when the two populations are separated by new infrastructure barriers, sink populations will loose the essential input of individuals from their sources and consequently face a rapid decline and ultimately extinction (Watkinson and Sutherland, 1995; and Figure



2.4). Despite this theoretical knowledge, sink and source dynamics are extremely difficult to recognise and quantify from simple field observations.

Figure 2.4 - Barrier effects on populations: (A) A metapopulation consists of a network of local populations that may vary in size and local dynamics, but are linked to each other through dispersal. Small local populations are more likely to go extinct than large populations, but the risks of this are minimised if they are well connected to surrounding populations from where they can be re-colonised; (B) Infrastructure construction causes a disturbance and loss of local populations within the network. In addition, infrastructure imposes a dispersal barrier that can prevent re-colonisation and isolate local populations from the rest of the metapopulation. If important source populations are cut off from the remaining sink populations, the entire metapopulation may be at risk of extinction.

2.4. PLANT AND ANIMAL MOVEMENTS

The movement of organisms is a fundamental property of life. Plants 'move' passively via natural (*e.g.* wind, water, and animals) or human (*e.g.* vehicles) vectors that transport their pollen or seeds (Verkaar, 1988; Wace, 1977). Few studies have been carried out to investigate the effect of infrastructure on plant movements, but there is evidence that weeds and many exotic plant species spread along infrastructure verges into adjacent habitats (see Section 3.3). Animals are more directly affected by infrastructure barriers, but to understand the problem and evaluate the conflict between the barriers and animal movements, it is necessary to recognise differences in the type of movements and the scale at which these occur (Verkaar and Bekker, 1991). Animals move within and between foraging areas, home ranges, regions and even continents. These movements are necessary for the daily survival of individuals as well as for the long-term persistence of populations. Broadly, four categories of movements can be distinguished (Figure 2.5 and Table 2-1).



Figure 2.5 - Four basic types of animal movements: (A) Foraging movements of an individual within a forest stand; (B) diurnal or commuting movements between forest patches within the home range of an individual; (C) dispersal movements (emigration and immigration) between local populations; (D) migratory movements between seasonal habitats by local populations. These movement types refer to different spatial and temporal scales, but may occur simultaneously in the landscape. (Drawings by Lars Jäderberg)

Movement	Features
Foraging	Made in order to access food sources within a habitat patch (Figure 2.5 A); they are small-scaled, convoluted and rather diffuse.
Diurnal or	Made regularly in the home range of an individual between different resources, <i>e.g.</i>
commuting	between breeding site, foraging areas, water and shelter (Figure 2.5 B); they are generally straight (often along guiding structures such as forest edges, hedgerows or rivers) and directed towards a goal (<i>e.g.</i> Saunders and Hobbs, 1991; Baudry and Burel, 1997).
Dispersal	Made when individuals leave their birthplace or parental home range in order to establish their own territory. Occurs once, or a few times, during the lifetime of an individual and serves to sustain local populations within a metapopulation (Figure 2.5 C). Little is known about patterns of dispersal but structures and corridors used in diurnal movements are often utilised.
Migratory	Cyclic, long-distance movements between seasonal habitats, often conducted by groups of individuals or even entire local populations. Represents an adaptation to a seasonally changing environment and is essential to the survival of many species. Animals often migrate along traditional paths used by previous generations for hundreds of years that cannot easily be changed in response to a new barrier (Figure 2.5 D).

Table 2-1 - Classification of Animal Movement 1	Patterns.
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Where infrastructure dissects a foraging, commuting, dispersal or migration route, animals will have to cross the barrier and encounter a higher risk of mortality from traffic impact (Verkaar and Bekker, 1991). Most traffic accidents involving deer, for instance, occur during the hours around sunset and sunrise, when the animals are moving to and from their preferred feeding sites (Groot Bruinderink and Hazebroek, 1996). Migratory species are especially vulnerable to the barrier and mortality effects associated with infrastructure. Amphibians, for example, migrate as entire populations between breeding ponds and terrestrial habitats and consequently suffer extreme losses due to traffic mortality (Sjögren-Gulve, 1994; Fahrig *et al.*, 1995). The migration of larger ungulates, such as moose (*Alces alces*) in northern Scandinavia (Sweanor and Sandegren, 1989; Andersen, 1991) and red deer (*Cervus elaphus*) in the Alps (Ruhle and Looser, 1991) also causes particular problems in relation to traffic safety.

Animal movements are an important consideration in wildlife management and conservation. Knowledge about the type and the extent of animal movement may help to increase traffic safety, reduce road mortality and/or find adequate places for mitigation measures such as fences and fauna passages (Putman, 1997; Finder *et al.*, 1999; Pfister, 1993; Keller and Pfister, 1997). Empirical data on animal movement is still limited and more field research is required in order to understand where, and how, artificial or semi-natural structures can be used to lead animals safely across infrastructure barriers.

2.5. CONNECTIVITY, CORRIDORS AND ECOLOGICAL NETWORKS

Habitat connectivity denotes the functional connection between habitat patches. It is a vital, species-specific property of landscapes, which enables the movement of an animal within a landscape mosaic (Baudry and Merriam, 1988; Taylor *et al.*, 1993). Connectivity is achieved when the distances between neighbouring habitat patches are short enough to allow individuals to cross easily on a daily basis. In fragmented landscapes, connectivity can be maintained through: i) a close spatial arrangement of small habitat patches serving as stepping-stones; ii) corridors that link habitats like a network and; iii) artificial measures such as fauna passages over roads and railways (Figure 2.6).

Hedgerows and field margins, wooded ditches, rivers, road verges and power-lines are all 'ecological corridors' (Merriam, 1991). These support and direct movements of wildlife, but may also serve as a refuge to organisms that are not able to survive in the surrounding landscape (see Section 3.3.2). Most of the empirical data on the use of ecological corridors by wildlife refers to insects, birds and small mammals (*e.g.* Bennett, 1990; Merriam, 1991; Fry, 1995; Baudry and Burel, 1997) (see also Chapter 5). Little is known yet about the use of these rather small-scale structures by larger mammals (Hobbs, 1992).



Figure 2.6 - Hedgerows and woody road verges ('Knicks') in northern Germany provide the only bush and tree vegetation available in the landscape. Together they create a network of green corridors on which many species in that area depend for shelter and food. Naturally, these corridors also have a strong impact on the movement of species that shy away from the open fields and pastures. (Photo by Andreas Seiler)

The re-creation of ecological corridors is envisioned as the most effective strategy against habitat fragmentation in Europe. Recently, the concept of an ecological infrastructure - promoting the movement of wildlife in an otherwise hostile environment (Van Selm, 1988), has become adopted as a conservation tool by landscape architects (Dramstad *et al.*, 1996), and road planners (Saunders and Hobbs, 1991; Seiler and Eriksson, 1997; Jongman, 1999). Strategic ecological networks, such as the NATURA 2000 network or the Pan-European Ecological Network (Bennett and Wolters, 1996; Bennett, 1999; Opstal, 1999) attempt to apply the concept on a European scale by seeking to link areas designated for nature conservation (Jongman, 1994). Considering these 'networks' in the planning of infrastructure may help to highlight critical bottlenecks in habitat connectivity and identify where special mitigation measures may be required in the future.

2.6. Scale and hierarchy

The concepts of scale and hierarchy are essential to the understanding of ecological pattern and processes in the landscape (Urban *et al.*, 1987; Golley, 1989; Wiens, 1989). *Scale* defines the spatial and temporal dimensions of an object or an event within a landscape; every species, process or pattern owns its specific scale (Figure 2.7). For the purposes of environmental impact assessment (EIA), the scale at which ecological studies are undertaken is a fundamental consideration which determines the type of mitigation solutions that are designed. If an EIA is limited to an individual habitat, the wider (and potentially more serious) impacts at the landscape scale will be overlooked. Conversely, if too large a scale is selected for study, small sites that together comprise important components of the ecological infrastructure in the landscape may be ignored.



Figure 2.7 - Domains of scale in space and time. Enlarging the scale shifts the focus towards higher organisational levels that reveal new processes and dynamics. Nb. large spatial scales refer to small scales in map dimension. (Combined from Wiens, 1989 and Haila, 1990)

Closely related to scale is the *hierarchical structuring of nature* in which any system at a given scale is composed of a number of sub-systems at smaller scales (O'Neill *et al.*, 1986). For example, a metapopulation is comprised of local populations, which in turn are made up of many individuals (Figure 2.8).



Figure 2.8 - Hierarchical layering in ecology. Food patches are nested in individuals' territories, which make up the habitat of a local population. In turn, these local populations make up metapopulations that together comprise the evolutionary deme of a species. At each hierarchical level (*i.e.* site, landscape, region, zone), the spatial entities are linked trough the movement of individuals. (Redrawn after Angelstam, 1992)

In order to predict the effects of habitat fragmentation in relation to ecological properties at a given level (*e.g.* for a population), both of the adjacent levels in the hierarchical system (*i.e.* individual and metapopulation) must be considered (Senft *et al.*, 1987; Bissonette, 1997). In terms of the application of this principle to infrastructure planning, a theoretical example is outlined below.

Imagine a new railway that is to be built through a forest. On a topographical map, the forest may comprise a rather homogeneous green area. From a biological point of view, however, the forest is home to numerous local populations of animals, such as beetles that live on old growth trees (see Figure 2.8), and it forms the territory of an individual lynx. A new railway through this landscape will affect the beetle primarily at the population level due to the destruction of their habitat and increased separation of local populations. Disturbance and barrier effects of the new infrastructure may drive some of the local populations to extinction,

but the metapopulation may still persist. For the lynx, the railway matters mostly at the individual level. Traffic increases mortality risk and the railway barrier may dissect the lynx's home range into smaller, unviable fragments. The lynx is a relatively rare species, in which the loss of one single individual can be significant to the population in a region.

Depending on the vulnerability of a species at regional scale, the effects on individuals or the population(s) have to be evaluated on a case-by-case basis and mitigation strategies designed accordingly. If studied solely from a local perspective, the importance of barrier and fragmentation effects are likely to be underestimated, because consequences to the populations will first become apparent at a larger spatial scale.

2.7. SUMMARY

This chapter has introduced some specific ecological concepts that are relevant to the better understanding of landscape pattern and process in infrastructure planning. For further reading on the presented topics, see Forman (1995), Bissonette (1997), Farina (1998), Sutherland (1998), or Jedicke (1994). The most important principles can be summarised as follows:

- The effects of infrastructure on nature cannot be evaluated solely from a local perspective; infrastructure planning must focus on the landscape scale.
- Habitat connectivity across the landscape is essential for ensuring the survival of wildlife populations. Connectivity can be provided by ecological 'green' corridors, 'stepping stones', or technical mitigation measures e.g. constructing a bridge between severed habitats.
- The impact of habitat fragmentation on wildlife is dependent on individual species and landscape characteristics. Where the impact is below a critical threshold, populations can be sustained, but beyond this threshold, seemingly small changes in the environment may cause unexpected and irreversible effects (e.g. the extinction of local populations). The larger the spatial scale concerned, the longer the time-lag until effects may be detectable.

Infrastructure planning needs to integrate both regional and local-scale issues. A hierarchical approach can help to identify the most important problems and their solutions at each planning level. People should 'think globally, plan regionally but act locally' (sensu Forman, 1995).

There is still a long way to go before ecological tools are fully developed and implemented in road planning, but since the problems and their solutions are universal, joint research and combined international efforts are required. Only through interdisciplinary work (between planners, civil engineers and ecologists) can effective tools for assessing, preventing and mitigating against the ecological effects of infrastructure, be developed and applied.

Landscape and wildlife ecology together provide a body of theories and methodologies for the assessment of ecological impacts such as habitat fragmentation. Empirical studies are, however, scarce and more research is needed to investigate the critical thresholds beyond which populations cannot be sustained. The construction and daily use of transportation infrastructure can result in wide ranging ecological impacts that need to be identified and addressed. The specific nature of these impacts is discussed in more detail in Chapter 3.

Chapter 3. Effects of Infrastructure on Nature

This chapter presents an overview of the major ecological impacts of infrastructure, with a particular focus on those effects that impact upon wildlife and their habitats. The focus of this chapter is on the primary effects of transportation infrastructure on nature and wildlife, as these are usually the most relevant to the transport sector. Secondary effects following the construction of new roads or railways, *e.g.* consequent industrial development, or changes in human settlement and landuse patterns, are dealt with in more depth in Chapter 5 (Section 5.5). For more discussion and data on secondary effects see Section 5.5. The physical presence of roads and railways in the landscape creates new habitat edges, alters hydrological dynamics, and disrupts natural processes and habitats. Maintenance and operational activities contaminate the surrounding environment with a variety of chemical pollutants and noise. In addition, infrastructure and traffic impose movement barriers to most terrestrial animals and cause the death of millions of individual animals per year. The various biotic and abiotic impacts operate in a synergetic way locally as well as at a broader scale. Transportation infrastructure causes not only the loss and isolation of wildlife habitat, but leads to a fragmentation of the landscape in a literal sense.

An increasing body of evidence relating to the direct and indirect ecological effects of transportation infrastructure on nature includes the comprehensive reviews of van der Zande *et al.* (1980); Ellenberg *et al.* (1981); Andrews (1990); Bennett (1991); Reck and Kaule (1993); Forman (1995); Spellerberg (1998); Forman and Alexander (1998); and Trombulak and Frissell (2000). Impressive, empirical data has also been presented in the proceedings of various symposia (*e.g.* Bernard *et al.*, 1987; Canters *et al.*, 1997; Pierre-LePense and Carsignol, 1999; Evink *et al.*, 1996, 1998 and 1999; and Huijser *et al.*, 1999). Bibliographies on the topic have been compiled by Jalkotzky *et al.* (1997), Clevenger (1998), Glitzner *et al.* (1999), and Holzang *et al.* (2000). Readers are encourages to consult these complementary sources for further information on the topics discussed in brief below.

3.1. PRIMARY ECOLOGICAL EFFECTS

Most empirical data on the effects of infrastructure on wildlife refers to primary effects measured at a local scale. Primary ecological effects are caused by the physical presence of the infrastructure link and its traffic. Five major categories of primary effects can be distinguished (Figure 3.1; see also: van der Zande *et al.* (1980); Bennett (1991); Forman (1995)):

- *Habitat loss* is an inevitable consequence of infrastructure construction. Besides the physical occupation of land, disturbance and barrier effects in the wider environment further decrease the amount of habitat that is suitable or available for wildlife.
- *Disturbance/Edge effects* result from pollution of the physical, chemical and biological environment as a result of infrastructure construction and operation. Toxins and noise affect a much wider zone than that which is physically occupied.

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- Mortality levels associated with traffic are steadily rising (millions of individuals are killed on infrastructure each year in Europe), but for most common species this, traffic mortality it is not considered as a severe threat to population survival. Collisions between vehicles and wildlife are also an important traffic safety issue, and attract wider public interest for this reason.
- *Barrier* effects are experienced by most terrestrial animals. Infrastructure restricts the animals' range, makes habitats inaccessible and can lead to isolation of the population.
- *Corridor* habitats along infrastructure can be seen as either positive (in already heavily transformed low diversity landscapes) or negative (in natural well conserved landscapes where the invasion of non native, sometimes pest species, can be facilitated).



Figure 3.1 - Schematic representation of the five primary ecological effects of infrastructure which together lead to the fragmentation of habitat. (Modified from van der Zande *et al.*, 1980)

The impact of these primary effects on populations and the wider ecosystem varies according to the type of infrastructure, landscape, and habitat concerned. Individual elements of infrastructure always form part of a larger infrastructure network, where synonymous effects with other infrastructure links, or with natural barriers and corridors in the landscape, may magnify the significance of the primary effects. The overall fragmentation impact on the landscape due to the combined infrastructure network may thus not be predictable from data on individual roads and railways. When evaluating primary (ecological) effects of a planned infrastructure project it is essential to consider both the local and landscape scales, and fundamentally, the cumulative impact of the link when it becomes part of the surrounding infrastructure network.

3.2. HABITAT LOSS

3.2.1. Land take

Motorways may consume more than 10 hectares (ha) of land per kilometre of road and as a large part of that surface is metalled/sealed it is consequently lost as a natural habitat for plants and animals. Provincial and local roads occupy less area per kilometre, but collectively they comprise at least 95% of the total road network and hence their cumulative effect in the landscape can be considerably greater. If all the associated features, such as verges, embankments, slope cuttings, parking places, and service stations etc. are included, the total area designated for transport is likely to be several times larger than simply the paved surface

of the road (Figure 3.2). In most European countries, the allocation of space for new infrastructure is a significant problem for landuse planning. It is not surprising therefore that landtake is a fundamental consideration in Environmental Impact Assessment (EIA) studies and forms a baseline for designing mitigation and compensation measures in modern infrastructure projects (OECD, 1994, see also Section 5.4.1).

The physical occupation of land due to infrastructure is most significant at the local scale; at broader scales it becomes a minor issue compared to other types of landuse. Even in rather densely populated countries such as The Netherlands, Belgium or Germany, the total area occupied by infrastructure is generally estimated to be less than 5-7% (Jedicke, 1994). In Sweden, where transportation infrastructure is sparser, roads and railways are estimated to cover about 1.5% of the total land surface whilst urban areas comprise 3% (Seiler and Eriksson, 1997; Sweden Statistics, 1999).



Figure 3.2 - Slope cuttings along a road in Spain. (Photo by Martí Pey/Minuartia Estudis Ambientals)

3.3. DISTURBANCE

The total area used for roads and railways is, however, not a reliable measure of the loss of natural habitat. The disturbance influence on surrounding wildlife, vegetation, hydrology, and landscape spreads much wider than the area that is physically occupied and contributes far more to the overall loss and degradation of habitat than the road body itself. In addition, infrastructure barriers can isolate otherwise suitable habitats and make them inaccessible for wildlife. The scale and extent of the spread of disturbances is influenced by many factors including: road and traffic characteristics, landscape topography and hydrology, wind patterns and vegetation type and cover. In addition, the consequent impact on wildlife and ecosystems also depends on the sensitivity of the different species concerned. To understand the pattern, more has to be learned about the different agents of disturbance.

Many attempts have been made to assess the overall width of the disturbance zone around infrastructure developments (Figure 3.3). Depending on which impacts have been measured, the estimations range from some tens of metres (Mader, 1987a) to several hundred metres (Reichelt, 1979; Reijnen *et al.*, 1995; Forman and Deblinger, 2000) and even kilometres (Reck and Kaule, 1993; Forman *et al.*, 1997). Thus, despite its limited physical extent,

transportation infrastructure is indeed one of the more important actors in the landscape and its total influence on landuse and habitat function has probably been widely underestimated. Forman (2000) estimated that transportation infrastructure in the USA directly affects an area that is about 19 times larger than the 1% of the USA land surface that is physically occupied.



Figure 3.3 - Disturbance effects spreading from a road into the surrounding landscape. The distance over which disturbances affect nature depends on topography, wind direction, vegetation and the type of disturbance. The width of the affected zone is likely to be larger than some hundred meters on average. (Redrawn after Forman *et al.*, 1997)

3.3.1. Physical disturbance

The construction of infrastructure affects the physical environment due to the need to clear, level, fill, and cut natural material. Construction work changes soil density, landscape relief, surface- and groundwater flows, and microclimate, and thus alters land cover, vegetation and habitat composition. Wetlands and riparian habitats are especially sensitive to changes in hydrology *e.g.* those caused by embankments (Findlay and Bourdages, 2000) and cuttings which may drain aquifers and increase the risk of soil erosion and extensive earthslides that have the potential to pollute watercourses with sediments (*e.g.* Forman *et al.*, 1997; Trombulak and Frissell, 2000). The canalisation of surface water into ditches can also significantly change water run-off and debris flows, and thereby modify disturbance regimes in riparian networks (Jones *et al.*, 2000).

The clearance of a road corridor changes microclimatic conditions: it increases light intensity, reduces air humidity, and creates a greater daily variation in air temperature. These changes are naturally strongest where the road passes through forested habitats *e.g.* Mader (1987a)

observed changes in forest microclimate up to 30 metres from the edge of a forest road. Artificial edges produced by road construction are usually sharp and can be compared to the new edges created by clear cutting in forests (Jedicke, 1994). The opening of the forest canopy will adversely affect the occurrence of forest interior species such as lichens or mosses, but can favour species adapted to open and edge habitats (*e.g.* Ellenberg *et al.*, 1981; Jedicke, 1994).

3.3.2. Chemical disturbance

Chemical pollutants such as road dust, salt, heavy metals, fertiliser nutrients, and toxins are agents which contribute towards the disturbance effect caused by transportation infrastructure. Most of these pollutants accumulate in close proximity to the infrastructure but, in some cases, direct effects on vegetation and fauna can be observed at distances over several hundreds of metres away (*e.g.* Evers, 1976; Santelmann and Gorham, 1988; Bergkvist *et al.*, 1989; Hamilton and Harrison, 1991; Reck and Kaule, 1993; Forbes, 1995; Angold, 1997).

Dust, mobilised from the infrastructure, is transported and deposited along verges and in nearby vegetation; epiphytic lichens and mosses in wetlands and arctic ecosystems are especially sensitive to this kind of pollution (*e.g.* Auerbach *et al.*, 1997). De-icing and other salts (*e.g.* NaCl, CaCl₂, KCl, MgCl₂) can cause extensive damage to vegetation (especially in boreal and alpine regions (Blomqvist, 1998) and to coniferous forests), contaminate drinking water supplies and reduce the pH-level in soil (which in turn increases the mobility of heavy metals) (Bauske and Goetz, 1993; Reck and Kaule, 1993). Heavy metals and trace metals *e.g.* Pb, Zn, Cu, Cr, Cd, Al (derived from petrol, de-icing salts, and dust) can accumulate in plant and animal tissues and can affect their reproduction and survival rates (Scanlon, 1987 and 1991). Traffic exhaust emissions contain toxins such as polycyclic aromatic hydrocarbons, dioxins, ozone, nitrogen, carbon dioxide, and many fertilising chemicals. Changes in plant growth and plant species diversity have been observed and directly attributed to traffic emissions in lakes (Gjessing *et al.*, 1984) and in heathland at a distance of over 200 metres away from the road (Angold, 1997).

3.3.3. Traffic noise

Although disturbance effects associated with noise are more difficult to measure and less well understood than those related to chemicals, it is considered to be one of the major factors polluting natural environments in Europe (Vangent and Rietveld, 1993; Lines *et al.*, 1994). Areas free from noise disturbance caused by traffic, industry or agriculture have become rare at a European scale and tranquillity is perceived as an increasingly valuable resource (Shaw, 1996). Although noise seldom has an immediate physiological effect on humans, long exposure to noise can induce psychological stress and eventually lead to physiological disorder (*e.g.* Stansfeld *et al.*, 1993; Lines *et al.*, 1994; Job, 1996; Babisch *et al.*, 1999). Whether wildlife is similarly stressed by noise is questionable (see Andrews, 1990), however, timid species might interpret traffic noise as an indicator of the presence of humans and consequently avoid noisy areas. For instance, wild reindeer (*Rangifer tarandus*) avoid habitats near roads or utilise these areas less frequently than would be expected from their occurrence in the adjacent habitat (Klein, 1971). Traffic noise avoidance is also well documented for elk, caribou and brown bear (Rost and Bailey, 1979; Curatolo and Murphy, 1986). However, whether this avoidance is related to the amplitude or frequency of traffic noise is not known.

Birds seem to be especially sensitive to traffic noise, as it directly interferes with their vocal communication and consequently their territorial behaviour and mating success (Reijnen and Foppen, 1994). Various studies have documented reduced densities of birds breeding near trafficked roads (*e.g.* Veen, 1973; Räty, 1979; van der Zande *et al.*, 1980; Ellenberg *et al.*, 1981; Illner, 1992; Reijnen and Foppen, 1994). Extensive studies on willow warblers (*Phylloscopus trochilus*) in The Netherlands showed the birds suffered lower reproductivity, lower average survival, and higher emigration rates close to trafficked roads (Foppen and Reijnen, 1994). Box 3.1 details some of the major studies that have contributed towards knowledge in this field.

It has been shown that environmental factors such as the structure of verge vegetation, the type of adjacent habitat, and the relief of the landscape will influence both noise spread and species density, and thus alter the amplitude of the noise impact (*e.g.* Reijnen *et al.*, 1997; Kuitunen *et al.*, 1998; Meunier *et al.*, 1999). If verges provide essential breeding habitats that are rare or missing in the surrounding landscape, species density along infrastructure may not necessarily be reduced, even though disturbance effects may reduce the environmental quality of these habitats (Laursen, 1981; Warner, 1992; Meunier *et al.*, 1999). Although strategic research regarding the disturbance thresholds of species in relation to infrastructure construction and operation is lacking, the species with the following attributes are considered to be most vulnerable to disturbance and development impacts (Hill *et al.*, 1997):

- large species;
- long-lived species;
- species with relatively low reproductive rates;
- habitat specialists;
- species living in open (e.g. wetland) rather than closed (e.g. forest) habitats;
- rare species;
- species using traditional sites; and
- species whose populations are concentrated in a few key areas (UK-SoA, 5.4.3).

3.3.4. Visual and other disturbance

The effects of traffic also include visual disturbance *e.g.* from artificial lighting or vehicle movement but these impacts do not generally receive as much attention as traffic noise or toxins. Artificial lighting has a conflicting effect on different species of fauna and flora: it can act as a valuable deterrent to deer and a readily accessible insect food supply to bats, but at the same time it can disrupt growth regulation in plants (Campbell, 1990; Spellerberg, 1998), breeding and behaviour patterns in birds (Lofts and Merton, 1968; Hill, 1992), bats (Rydell, 1992), nocturnal frogs (Buchanan, 1993), and moth populations (Frank, 1990; Svensson and Rydell, 1998). A study on the influence of road lights on a black-tailed godwit (*Limosa limosa*) population in The Netherlands, for example, indicated that the breeding density of this species was significantly reduced in a zone of 200 to 250 metres around the lights (De Molenaar *et al.*, 2000).

Certain types of road lights, such as white (mercury vapour) street lamps are especially attractive to insects, and therefore also to aerial-hawking bat species such as pipistrelles (*Pipistrellus pipistrellus*) (Rydell, 1992; Blake *et al.*, 1994). This increases the exposure of bats to traffic and may entail increased mortality due to collisions with vehicles. Furthermore, lit roads can constitute linear landscape elements, which bats may use to navigate in open areas (UK-SoA).

Box 3.1 - Studies on the effect of traffic noise on breeding birds

Between 1984 and 1991, the Institute for Forest and Nature Research in The Netherlands has carried out extensive studies of the effect of motorways and roads with traffic intensities between 5,000 and 60,000 vehicles a day on populations of breeding birds (Reijnen *et al.*, 1992; Reijnen, 1995). Two types of landscape, forest (Reijnen *et al.*, 1995a) and open grassland (Reijnen *et al.*, 1996) were compared. For 33 of the 45 forest species and 7 of 12 open grassland species, a road traffic effect was established and bird densities declined where the traffic noise exceeded 50 decibels (dbA). Birds in woodland reacted at noise levels of only 40 dbA. It was concluded that road traffic has an effect on the total density of all species and that there are clear indications that traffic noise is the main disturbing factor responsible for reduced densities of breeding birds near roads.

Based on the observed relationship between noise burden and bird densities, Reijnen, Veenbaas and Foppen (1995) proposed a simple model predicting the distance over which breeding bird populations might be affected by traffic noise (Figure 3.4). According to this model, roads with a traffic volume of 10,000 vehicles per day and a traffic speed of 120 km/h, passing through an area with 70% woodland, would significantly affect bird densities at distances between 40 and 1,500 m. When the model is applied to the entire area of The Netherlands, it suggests that at least 17% of bird habitats are affected by traffic noise (Reijnen *et al.*, 1995b).



Figure 3.4 - Schematic representation of the impact of traffic noise on breeding bird populations in The Netherlands. When the noise load exceeds a threshold of between 40 and 50 dBA, bird densities may drop significantly. The sensitivity to noise and thus the threshold is different between species and between forest and open habitats. (From Reijnen, Veenbaas and Foppen, 1995)

Helldin and Seiler (2001) tested the predictions of Reijnen *et al.* (1995a) model for Swedish landscapes and found that the expected reduction in breeding bird densities could not be verified. On the contrary, some species even tended to increase in densities towards the road. It was concluded that the Dutch model might not be directly applicable in other countries and that habitat changes as a consequence of road construction under some circumstances could override the negative effects of traffic noise on the surroundings (S-SoA, 5.4.3). Species are negatively affected due to the artificial lighting upsetting their natural biological systems which are reliant on day length, and disturbing their spatial orientation and diurnal activity patterns. It is therefore possible that mitigation measures will also have conflicting effects on different species. From the studies that have been carried out, the following basic principles for reducing the impact of road lighting are suggested:

- Avoid lighting on roads crossing natural areas; and
- Use methods of lighting which are less alluring, especially for insects.

The movement of vehicles (probably in combination with noise) can also alter behaviour and induce stress reactions in wildlife. Madsen (1985), for instance, observed that geese foraging near roads in Denmark were more sensitive to human disturbance than when feeding elsewhere. Reijnen *et al.* (1995a) did not observe any effect of the visibility of moving cars on breeding birds, however, Kastdalen (*pers. comm.*) reported that moose (*Alces alces*) approaching a fauna passage under a motorway in Norway ran off as large trucks passed overhead. Heavy trucks and, more especially, high-speed trains produce intensive, but discontinous noise, vibration and visual disturbance which has the effect of frightening many mammals and birds. It is documented that many larger mammals avoid habitats in the vicinity of trafficked roads and railways (*e.g.* Klein, 1971; Rost and Bailey, 1979; Newmark *et al.*, 1996), but this avoidance results from many different interacting factors, amongst which noise and visual disturbance from vehicles comprise a small part.

3.3.5. Conclusions

Artificial lighting, traffic noise, chemical pollutants, microclimatic and hydrological changes, vibration and movement are just a few sources of disturbance that alter the habitats adjacent to infrastructure. In many situations, such disturbances are probably of marginal importance to wildlife, and many animals habituate quickly to constant disturbance (as long as they do not experience immediate danger). This does not imply, however, that disturbance should not be considered during the EIA process. On the contrary, because measures to mitigate against these types of disturbance are usually simple and inexpensive to install, they can easily be considered and integrated during the planning and design process. Many of the studies cited above were not specifically designed to directly investigate the disturbance effect of infrastructure, nor to inform the development of tools for impact evaluation or mitigation. However, to assess the width and intensity of the road-effect zone, research is needed that specifically addresses the issue of the spread of disturbance and the effect thresholds for individual species. Until there is a better understanding of such issues, the precautionary principle should be applied in all cases to prevent unnecessary negative effects.

3.4. CORRIDOR FUNCTION

Planted areas adjacent to infrastructure are highly disturbed environments, often hostile to many wildlife species, yet they can still provide attractive resources such as shelter, food or nesting sites, and facilitate the spread of species. In heavily exploited landscapes, infrastructure verges can provide valuable refuges for species that otherwise could not survive. Verges, varying in width from a few metres up to several tens of metres, are multipurpose areas, having to fulfil technical requirements such as providing free sight for drivers thus promoting road safety, and screening the road from the surrounding landscape. Typically, traffic safety requires that the vegetation adjacent to roads is kept open and grassy but farther away from the road, verges are often planted with trees and shrubs for aesthetic reasons, or to buffer the spread of salt and noise (Figure 3.5). Balancing technical and biological interests in the design and management of verges is a serious challenge to civil engineering and ecology. It offers a great opportunity for the transport sector to increase and protect biodiversity at large scale (Mader, 1987b; Van Bohemen *et al.*, 1991; Jedicke, 1994).



Figure 3.5 - Verges can vary considerably between different landscapes and countries. Left: A motorway in southern Sweden consisting only of an open ditch. Toxins and salt from the road surface can easily spread onto the adjacent agricultural field. Right: A highway in Germany. Densely planted shrubs and trees along roads provide potential nesting sites for birds and screen the road and its traffic from the surrounding landscape. (Photos by A. Seiler)

3.4.1. Verges as habitat for wildlife

Numerous inventories indicate the great potential of verges to support a diverse range of plant and animal species (*e.g.* Hansen and Jensen, 1972; Mader *et al.*, 1983; Van der Sluijs and Van Bohemen, 1991; Sjölund *et al.*, 1999). Way (1977) reported that verges in Great Britain supported 40 of the 200 native bird species, 20 of 50 mammalian, all 6 reptilian species, 5 of 6 amphibian, and 25 of the 60 butterfly species occurring in the country. In areas, where much of the native vegetation has been destroyed due to agriculture, forestry or urban development, verges can serve as a last resort for wildlife (Loney and Hobbs, 1991). Many plant and animal species in Europe that are associated with traditional (and now rare) grassland and pasture habitats, may find a refuge in the grassy verges along motorways and railways (Sayer and Schaefer, 1989; Melman and Verkaar, 1991; Ihse, 1995; Auestad *et al.*, 1999). Shrubs and trees can provide valuable nesting sites for birds and small mammals (Adams and Geis, 1973; Laursen, 1981; Havlin, 1987; Meunier *et al.*, 1999) and also offer food and shelter for larger species (Klein, 1971; Rost and Bailey, 1979).

Other elements of the infrastructure itself can also provide attractive, yet sometimes hazardous, habitat for wildlife. For instance, stone walls and drainage pipes under motorways in Catalonia, Northeast Spain, are often populated by lizards and common wall geckos (*Tarentola mauritanica*) (Rosell and Rivas, 1999). Cavities in the rocky embankments of railways may be used as shelter and breeding sites by lizards (Reck and Kaule, 1993) and bats may find secure resting sites underneath bridges (Keeley and Tuttle, 1999). However, caution needs to be given to the inherent hazards associated with these structures. In the UK, for example, drainage pipes are recognised as representing a significant mortality risk to reptiles (Tony Sangwine, *pers comm*.). Careful design, management and maintenance of these structures is required in order to minimise the potentially negative impacts on the wildlife

utilizing them. The first objective should be to identify which engineering elements may be of benefit to which species, and the second to determine how this benefit can be maximised without compromising the primary function of the structure.

Many wildlife species can benefit from verges if they provide valuable resources that are rare or missing in the surrounding landscape. However, it is unlikely that these human-made habitats will develop the ecological value of comparable natural habitat types found some distance from the infrastructure. The composition of species found in transportation infrastructure verges is generally skewed towards a higher proportion of generalists and pioneers that can cope with high levels of disturbance (Hansen and Jensen, 1972; Adams and Geis, 1973; Niering and Goodwin, 1974; Douglass, 1977; Mader *et al.*, 1983; Blair, 1996). It is not surprising that species, which regularly visit road corridors to forage or nest, feature frequently in traffic mortality statistics (see Section 3.5). In this respect, infrastructure corridors may act as an ecological trap, outwardly offering favourable habitat conditions but with the hidden high risk of mortality. When designing and managing verges, it is therefore advisable to consider the risk of creating an ecological trap that may kill more species than it sustains.

3.4.2. Verges as movement corridors for wildlife

As well as providing a habitat for wildlife, verges may also serve as a conduit for species movement (active or passive) like 'natural' corridors in the landscape (see Section 2.4). In The Netherlands, bank voles (Clethrinomys glareolus) have colonised the Zuid-Beveland peninsula after moving along wooded verges of railways and motorways (Bekker and Mostert, 1998). Getz et al. (1978) documented that meadow voles (Microtus pennsylvanicus) dispersed over about 100 km in six years along grassy verges in Illinois, USA. Kolb (1984) and Trewhella and Harris (1990) observed that the movement of foxes (Vulpes vulpes) into the Edinburgh area of the UK was strongly influenced by the presence and direction of railway lines. Badgers living in the city of Trondheim, Norway, are known to use riverbanks and road verges to move within the city (Bevanger, pers. comm.). The actual surface of the infrastructure (mainly small roads with little traffic) may also be used as pathways by larger mammals. Vehicle and human movement along the infrastructure may also serve as a vector for plants, seeds or small, less mobile animals (Schmidt, 1989; Bennett, 1991). For instance, Wace (1977) found seeds of 259 plant species in the sludge of a car-washer in Canberra, Australia, some of which derived from habitats more than 100 km away. This accidental transport of seeds may offer an explanation for the high proportion of exotic and weed species found along verges (Mader et al., 1983; Tyser and Worley, 1992; Ernst, 1998) that are considered a severe threat to native flora (Usher, 1988; Spellerberg, 1998).

It is clear that infrastructure verges can facilitate animal movement and enable the spread of plants and other sessile species. It may therefore seem feasible to integrate infrastructure corridors into the existing (natural) ecological network (Figure 2.6). However, several important characteristics distinguish verges from 'natural' corridors and may hamper a successful linkage between technical and ecological infrastructure (Mader 1978b; Mader *et al.*, 1990). Habitat conditions (particularly microclimatic and hydrological) vary considerably within verges and infrastructure networks have intersections where animals face a higher risk of traffic mortality than if they had travelled along another natural corridor in the landscape (Madsen *et al.*, 1998; Huijser *et al.*, 1998).

Also, the predation pressure within verges may be increased compared to the surrounding habitat, because carnivores are attracted to traffic casualties as a food source.

Thus, the overall corridor effect is ambiguous. Verges may provide valuable habitats for wildlife, but primarily for less demanding, generalist species that are tolerant of disturbance and pollution and are resilient to the increased mortality risk associated with the traffic. Verges can support wildlife movements, but also serve as a source of 'unwanted' or alien species spreading into the surrounding habitats. The overall corridor function of infrastructure verges will most likely be influenced by the ecological contrast between the vegetation/structure in the corridor and the surrounding habitat (Figure 3.6). To better understand this complexity and give practical advice to road planners, more empirical studies are needed.



Figure 3.6 - The corridor function differs with respect to the surrounding landscape: A) Open, agricultural landscapes: richly vegetated verges can provide a valuable habitat for wildlife and facilitate movement. B) Forested landscapes: open and grassy verges introduce new edges and can increase the barrier effect on forest interior species. C) Verges may also serve as sources of species spreading into new habitats or re-colonising vacant areas. (Modified from Mader, 1987b)

3.5. FAUNA CASUALTIES

3.5.1. The phenomenon

Road mortality is probably the most widely acknowledged effect of traffic on animals, as carcasses are a common sight along trafficked roads (Figure 3.7). The number of casualties appears to be constantly growing as traffic increases and infrastructure expands (Stoner 1925; Trombulak and Frissell, 2000). Forman and Alexander (1998) concluded that 'sometime during the last three decades, roads with vehicles probably overtook hunting as the leading direct human cause of vertebrate mortality on land'. The scale of the problem is illustrated by the numbers of known road kills (see Section 5.3 and Table 5.7).



Figure 3.7 - Wildlife casualties – a common view along roads and railways. (Photos by H. De Vries and C. Rosell)

The quantity of road kills is such that collisions between vehicles and wildlife comprise a growing problem not only for species conservation and game management, but also for traffic safety, and the private and public economy (Harris and Gallagher, 1989; Hartwig, 1993; Romin and Bissonette, 1996; Putman, 1997). In most countries, traffic safety is the driving force behind mitigation efforts against fauna casualties (see Chapter 8) and although human fatalities are a relatively rare outcome in wildlife-vehicle collisions, the number of injured people and the total economic costs, including damage to vehicles, can be substantial. Police records in Europe (excluding Russia) suggest more than half a million ungulate-vehicle collisions per year, causing a minimum of 300 human fatalities, 30,000 injuries, and a material damage of more than 1 billion Euro (Groot Bruinderink and Hazebroek, 1996). From an animal welfare point of view, there is also concern about road casualties: many animals that are hit by vehicles are not immediately killed, but die later from injuries or shock. Hunters complain about the increasing work to hunt down injured game (Swedish Hunters Association, *pers. comm.*) and train drivers in northern Sweden complain about the unpleasant experience of colliding with groups of reindeer and moose (Åhren and Larsson, 1999).

3.5.2. Ecological significance of wildlife-traffic collisions

Evaluating the ecological importance of road mortality for a species involves considering the species' population size and recruitment rate. Large numbers of casualties of one species may not necessarily imply a threat to the survival of that species, but rather indicate that it is abundant and widespread. For many common wildlife species, such as rodents, rabbits, foxes, sparrows, or blackbirds, traffic mortality is generally considered insignificant, accounting only for a small portion (less than 5%) of the total mortality (Haugen, 1944; Bergmann, 1974; Schmidley and Wilkins, 1977; Bennett, 1991; Rodts *et al.*, 1998; see also Table 5.7). Even for red deer (*Cervus elaphus*), roe deer (*Capreolus capreolus*) or wild boar (*Sus scrofa*), traffic

mortality generally accounts for less than 5% of the annual spring populations in Europe (Groot Bruinderink and Hazebroek, 1996). In contrast to natural predation, traffic mortality is non-compensatory, and the kill rate is independent of density. This implies that traffic will kill a constant proportion of a population and therefore affect rare species most significantly. In general, species that occur in small isolated populations, and those which require large extensive areas for their home ranges, or exert long migratory movements, are especially sensitive to road mortality. Indeed, for many endangered or rare species around the world, traffic is considered as one of the most important sources of mortality (Harris and Gallagher, 1989).

3.5.3. Factors that influence the occurrence of wildlife-traffic collisions

There are various factors that determine the risk of animal-vehicle collisions (Figure 3.8). The numbers of collisions generally increase with traffic intensity and animal activity and density. Temporal variations in traffic kills can be linked to biological factors which determine the species' activity *e.g.* the daily rhythm of foraging and resting, seasons for mating and breeding, dispersal of young, or seasonal migration between winter and summer habitats (Van Gelder, 1973; Bergmann, 1974; Göransson *et al.*, 1978; Aaris-Sorensen, 1995; Groot Bruinderink and Hazebroek, 1996). Changes in temperature, rainfall or snow cover can also influence the occurrence and timing of accidents (Jaren *et al.*, 1991; Belant, 1995; Gundersen and Andreassen, 1998).



Figure 3.8 - Factors influencing the number of wildlife traffic accidents.

Roadkills seem to increase with traffic intensity to an optimum point, after which they level off. It seems that very high traffic volumes, noise and vehicle movements have the effect of deterring many animals, hence mortality rates do not increase further with higher traffic flows (Oxley *et al.*, 1974; Berthoud, 1987; Van der Zee *et al.*, 1992; Clarke *et al.*, 1998; see Figure 3.10). The occurrence of mitigation measures such as fences or passages and the programme of verge management clearly affects the local risk of accidents. The clearance of infrastructure verges of deciduous vegetation, for instance, has proven to reduce the number of moose (*Alces alces*) casualties in Scandinavia by between 20% and 50% (Lavsund and Sandegren, 1991; Jaren *et al.*, 1991). On the other hand, where verges provide attractive

resources to wildlife, the risk of vehicle-animal collisions is likely to be increased (Feldhamer *et al.*, 1986; Steiof, 1996; Groot Bruinderink and Hazebroek, 1996).

Spatial pattern in road kills clearly depends on animal population density and biology, habitat distribution and landscape structure, but also on road and traffic characteristics (Puglisi *et al.*, 1974'; Ashley and Robinson, 1996, Finder *et al.*, 1999). In species with limited mobility and specific habitat requirements, such as many amphibians, it can be relatively simple to identify potential conflict areas. Most amphibian casualties occur during a short period in spring, when the animals migrate to and from their breeding ponds and are concentrated where roads dissect the migration routes (van Gelder, 1973). Roads that pass close to breeding ponds, wetlands and the animals' foraging habitats, are likely to cause a much greater kill rate than roads outside the species' migratory range *i.e.* about 1 km (see Vos and Chardon, 1998; Ashley and Robinson, 1996).

Other species, especially larger mammals, depend less on specific habitat types and utilise the landscape at a broader scale, which makes it more difficult to locate possible collision 'hotspots' (Madsen *et al.*, 1998). However, where favourable habitat patches coincide with infrastructure, or where roads intersect other linear structures in the landscape (*e.g.* hedgerows, watercourses, and other (minor) roads and railways), the risk of collisions is usually increased (Puglisi *et al.*, 1974; Feldhamer *et al.*, 1986; Kofler and Schulz, 1987; Putman, 1997; Gundersen *et al.*, 1998; Lode, 2000). For example, collisions with white-tailed deer (*Odocoileus virginianus*) in Illinois are associated with intersections between roads and riparian corridors, and public recreational land (Finder *et al.*, 1999). Traffic casualties amongst otters (*Lutra lutra*) are most likely to occur where roads cross over watercourses (Philcox *et al.*, 1999). Road-killed hedgehogs (*Erinaceus europaeus*) in The Netherlands are often found where roads intersect with railways (Huijser *et al.*, 1998). Also foxes and roe deer (*Capreolus capreolus*) in Denmark are more often found near intersections than elsewhere along roads (Madsen *et al.*, 1998).

The different factors influencing wildlife-traffic accidents must be fully understood before any local need for mitigation can be evaluated, and effective measures designed and constructed (Romin and Bissonette, 1996; Putman, 1997). GIS-based analysis of traffic kills and wildlife movements, in relation to roads and landscape features, may provide the necessary insight to enable predictive models for impact assessment and the localisation of mitigation measures to be developed and applied (Gundersen *et al.*, 1998; Finder *et al.*, 1999; see also Section 6.4).

3.6. BARRIER EFFECT

3.6.1. The components of the barrier effect

Of all the primary effects of infrastructure, the barrier effect contributes most to the overall fragmentation of habitat (Reck and Kaule, 1993; Forman and Alexander, 1998). Infrastructure barriers disrupt natural processes including plant dispersal and animal movements (Forman *et al.*, 1997). The barrier effect on wildlife results from a combination of disturbance and avoidance effects (*e.g.* traffic noise, vehicle movement, pollution, and human activity), physical hindrances, and traffic mortality that all reduce the number of movements across the infrastructure (Figure 3.9). The infrastructure surface, gutter, ditches, fences, and embankments may all present physical barriers that animals cannot pass. The clearance of the infrastructure corridor and the open verge character creates habitat conditions that are

unsuitable or hostile to many smaller species (see Section 3.3.1). Most infrastructure barriers do not completely block animal movements, but reduce the number of crossings significantly (Merriam *et al.*, 1989). The fundamental question is thus: how many successful crossings are needed to maintain habitat connectivity?



Figure 3.9 - The barrier effect of a road or railway results from a combination of disturbance/deterrent effects, mortality and physical hindrances. Depending on the species, the number of successful crossings is but a fraction of the number of attempted movements. Some species may not experience any physical or behavioural barrier, whereas others may not try to even approach the road corridor. To effectively mitigate the barrier effect, the relative importance of the inhibiting factors on individual species must be established.

The barrier effect is a non-linear function of traffic intensity, which along with vehicle speed appear to have the strongest influence on the barrier effect. Infrastructure width, verge characteristics, the animals' behaviour and its sensitivity to habitat disturbances are also key factors (Figure 3.10). With increasing traffic density and higher vehicle speed, mortality rates usually increase until the deterrent effect of the traffic prevents more animals from getting killed (Oxley *et al.*, 1974; Berthoud, 1987; Kuhn, 1987; Van der Zee *et al.* 1992; Clarke *et al.* 1998). Exactly when this threshold in traffic density occurs is yet to be established but Müller and Berthoud (1997) propose five categories of infrastructure/traffic intensity with respect to the barrier impact on wildlife:

- Local access and service roads with very light traffic: can serve as partial filters to wildlife movements; may have a limited barrier impact on invertebrates and eventually deter small mammals from crossing the open space; larger wildlife may benefit from these roads as corridors or conduits.
- Railways and minor public roads with traffic below 1,000 vehicles per day: may cause incidental traffic mortality and exert a stronger barrier/avoidance effect on small species, but crossing movements still occur frequently.
- Intermediate link roads with up to 5,000 vehicles per day: may already represent a serious barrier to certain species; traffic noise and vehicle movement are likely to have a major deterrent effect on small mammals and some larger mammals meaning the increase in the overall barrier impact is not proportional to the increase in traffic volume.

- Arterial roads with heavy traffic between 5,000 and 10,000 vehicles per day: represent a significant barrier to many terrestrial species, but due to the strong repellence effect of the traffic, the number of roadkills remains relatively constant over time; roadkills and traffic safety are two major issues in this category.
- Motorways and highways with traffic above 10,000 vehicles per day: impose an impermeable barrier to almost all wildlife species; dense traffic deters most species from approaching the road and kills those that still attempt to cross.



Figure 3.10 - Theoretical model illustrating the relationship between traffic intensity and the barrier effect: with increasing traffic, the number of roadkills increases in a linear fashion until noise and vehicle movements repel more animals from attempting to cross the road; at very high traffic volumes, the total mortality rate could decrease until the barrier effect reaches 100% *i.e.* preventing all crossings. (Redrawn from Müller and Berthoud, 1997)

3.6.2. Evidence from field studies

Transportation infrastructure inhibits the movement of practically all terrestrial animals, and many aquatic species: the significance of the barrier effect varies between species. Many invertebrates, for instance, respond significantly to differences in microclimate, substrate and the extent of openness between road surface and road verges: high temperatures, high light intensity and lack of shelter on the surface of paved roads have been seen to repel Lycosid spiders and Carabid beetles (Mader 1988; Mader *et al.*, 1990). Land snails may dry out or get run over while attempting to cross over a paved road (Baur and Baur, 1990). Also amphibians, reptiles, and small mammals may be sensitive to the openness of the road corridor, the road surface and traffic intensity (Joule and Cameron, 1974; Kozel and Fleharty, 1979; Mader and Pauritsch, 1981; Swihart and Slade, 1984; Merriam *et al.*, 1989; Clark *et al.*, 2001). Even birds can be reluctant to cross over wide and heavily trafficked roads (Van der Zande *et al.*, 1980). Semi-aquatic animals and migrating fish moving along watercourses are often be inhibited by bridges or culverts that are too narrow (Warren and Pardew, 1998).

Most empirical evidence for the barrier effect derives from capture-recapture experiments on small mammals. For example, Mader (1984) observed that a 6 m wide road with 250 vehicles/hour completely inhibited the movement of 121 marked yellow-necked mice (*Apodemus flavicollis*) and bank voles (*Clethrionomys glareolus*) (see Figure 3.11). Similarly,

Richardson *et al.* (1997) found that mice and voles were reluctant to cross paved roads wider than 20-25 m although they did move along the road verge. Oxley *et al.* (1974) documented that white-footed mice (*Peromyscus leucopus*) would not cross over highway corridors wider than 30 m although they frequently crossed over smaller and only lightly trafficked forest roads.



Figure 3.11 - Mobility diagram illustrating animal movements along and across a railway and road, based on capture-recapture data of: (left) carabid beetles (redrawn from Mader *et al.*, 1990); and (right) small mammals. (Redrawn from Mader, 1984)

For larger animals, roads and railways do not represent a physical barrier, unless they are fenced or their traffic intensity is too high. Most mammals, however, are sensitive to disturbance by humans and scent, noise and vehicle movement may deter animals from approaching the infrastructure corridor. For example, Klein (1971) and Curatolo and Murphy (1986) observed a strong avoidance of roads by feral reindeer (but not by domestic reindeer) and Rost and Bailey (1979) reported that mule deer (*Odocoileus hemionus*) and elk (*Cervus canadensis*) avoided habitats closer than around 100 m to trafficked roads.

However, to what extent this avoidance effect reduces the number of successful or attempted movements across roads is not clear. More data is required on the actual movements (spatial and temporal) of larger mammals in relation to infrastructure in order to judge the inhibitory effect of roads and traffic.

3.6.3. Consequences at a population level

When do infrastructure barriers really become a problem for wildlife conservation? How much permeability is needed to maintain sufficient habitat connectivity? How large a barrier effect can be tolerated by individual species and populations? To answer these questions, the consequences at population level must be considered. Depending on the number of successful crossings relative to the size of the population, the barrier effect can be significant to population dynamics, demographic or genetic properties. If the species does not experience a significant barrier effect and individuals still move frequently across the road, the dissected populations will continue to function as one unit. If the exchange of individuals is reduced but not completely inhibited, the populations may diverge in demographic characters, *e.g.* in terms of density, sex ratio, recruitment and mortality rate. Also genetic differences may emerge, as the chance for mating with individuals from the other side of the infrastructure barrier may be reduced. These changes may not necessarily pose a threat to the dissected populations; except for sink populations dependent on steady immigration for continued survival (see Section 2.3). If the barrier effect is even stronger, the risk of inbreeding effects and local extinctions will increase rapidly.

Evidence of the effect on population genetics derives from studies on rodents and amphibians. For example, Reh and Seitz (1990) observed effects of inbreeding, in the form of reduced genetic diversity, in small populations of the common frog (*Rana temporaria*) that were isolated by roads over many years. Merriam *et al.* (1989) found indications of genetic divergence in small-mammal populations separated by minor roads. However, populations dissected by one single barrier may not automatically suffer from inbreeding depression, unless they are critically small or do not have contact with other more distant populations in the landscape. To evaluate the consequences of a new infrastructure barrier, the combined isolation effects of all the existing surrounding infrastructure and other natural and artificial barriers must be considered. The denser the infrastructure network and the more intense its traffic, the more likely it will cause significant isolation of local populations. By definition, small isolated populations (particularly of rare and endemic species) are more sensitive to barrier effects and isolation than populations of abundant and widespread species. Species with large area requirements and wide individual home ranges will more frequently need to cross over road barriers than smaller and less mobile species.

It is the combination of population size, mobility, and the individuals' area requirements that determines a species' sensitivity to the barrier impact of infrastructure (Verkaar and Bekker, 1991). A careful choice between alternative routes for new infrastructure may thus help to prevent the dissection of local populations of small species, but cannot reduce the barrier effect for larger, wide roaming species. In most cases, technical/physical measures, such as fauna passages or ecoducts, will be required to mitigate against barrier impacts and re-establish habitat connectivity across the infrastructure.

3.7. FRAGMENTATION

The previous discussions show that the total impact of roads and railways on wildlife cannot be evaluated without considering a broader landscape context. Roads and railways are always part of a wider network, where synergetic effects with other infrastructure links occur, which cause additional habitat loss and isolation. Studies on the cumulative effects of fragmentation caused by transportation infrastructure must address larger areas and cover longer time periods than studies that simply address the primary effects of a single road or railway link. Evaluating the degree of fragmentation due to infrastructure is not a simple task. The significance of fragmentation is highly species-specific and dependent on the amplitude of barrier and disturbance effects, the diversity and juxtaposition of habitats within the landscape, and the size of the unfragmented areas between infrastructure links (*i.e.* the density of infrastructure). Forman *et al.* (1997) suggested the use of infrastructure density as a simple but straightforward measure of fragmentation (Figure 3.12). This measure could be improved by adding information on traffic density, speed, infrastructure width and design.



Figure 3.12 - Infrastructure causes a loss and degradation of habitat due to disturbance effects (grey corridors) and isolation. With increasing infrastructure density, areas of undisturbed habitat (white) are reduced in size and become inaccessible. Remnant fragments of suitable habitat may eventually become too small and isolated to prevent local populations from going extinct. The critical threshold in road density is species-specific, but will also depend on landscape and infrastructure characteristics.

Several studies have described critical thresholds in road density for the occurrence of wildlife species in the landscape. For example, Mladenoff *et al.* (1999) observed that wolves and mountain lions did not sustain viable populations in regions of Minnesota, USA with road densities above 0.6 km/km² (Thiel, 1985; Van Dyke *et al.*, 1986). Also, the presence of other large mammals in the USA such as elk, moose and grizzly bear, appears to be negatively influenced as road densities increase (Holbrook and Vaughan, 1985; Forman *et al.*, 1997).

The observed fragmentation effect may however not be associated with the direct impact of infrastructure and traffic, but rather with the increased access to wildlife areas that roads in particular (especially forest roads) offer hunters and poachers (Holbrook and Vaughan, 1985; Gratson and Whitman, 2000). In Europe, areas remote from roads or with only low road density, low traffic volumes, and a high proportion of natural vegetation, are considered as core areas in the ecological network (e.g. Jongman, 1994; Bennett, 1997). Determining how much undeveloped habitat is needed and how large the infrastructure-free landscape fragments need to be to ensure a given species survival is a task for future research. Clearly, the best option to counteract the fragmentation process is the reclamation of nature areas for wildlife through the removal of roads, or by permanent or temporary road closure. Road closure helps to reduce motorised access to wildlife habitat and enlarges undisturbed core areas, yet the physical barrier and its edge effects still remain. The physical removal of roads is the ultimate solution. In some countries, such as on federal land in the USA, attempts are being made to integrate road removal as a part of the Grizzly Bear Conservation Program (see Evink et al., 1999; Wildlands CPR, 2001). To ensure the survival of grizzlies in the core areas of their distribution, it has been suggested to establish road-free habitats of at least 70% of the size of an average female home range. In regions designated for grizzly bear conservation and where road densities are higher than that required for the secure habitats, it is recommended that roads should consequently be removed.

In Europe, temporary closure of (local) roads is an action primarily applied in order to maximise the protection of seasonally migrating amphibians (Dehlinger, 1994). Applying speed limits on local roads can also offer a simple tool for changing traffic flows and reducing disturbance and mortality impacts in wildlife areas. In situations where roads cannot be removed or closed, or traffic reduced, technical mitigation measures such as fauna passages and ecoducts may be necessary to minimise fragmentation and reconnect wildlife habitats (*e.g.* DWW, 1995).

3.8. SUMMARY

In this chapter some of the major literature on the ecological effects of infrastructure has been reviewed. There is a growing concern about habitat fragmentation caused by roads and railways all around the world. The increasing demand for avoidance and mitigation makes it clear that there is still much to be understood before the cumulative potential impacts can be assessed in an efficient and practical way. A considerable amount of research has been carried out already, yet many of the studies are descriptive, dealing with problems of individual roads or railways, but without considering the more strategic issues integral in the planning of ecologically friendly infrastructure.

How much habitat is actually lost due to construction and disturbance effects of infrastructure? How wide is the impact zone along roads and how does the width of this zone change with traffic intensity and type of surrounding habitat? How can transportation infrastructure be integrated into the 'ecological' infrastructure in the landscape without causing an increase in the risk of animal-vehicle collisions? Where and when are mitigation measures against road wildlife mortality necessary or affordable? How much infrastructure is too much in areas designated for wildlife? What are the ecological thresholds that must not be surpassed and how can the best use be made of the potential in a road or railway project to improve the current situation?

Finding answers to these questions is a challenge to landscape ecologists, biologists and civil engineers alike (Forman, 1998; Cuperus *et al.*, 1999). To develop effective guidelines and tools for the planning of infrastructure, research needs to be focussed on ecological processes and patterns, using experiments and simulation models to identify critical impact thresholds. Empirical studies are necessary to provide the basic data that will help to define evaluation criteria and indices. Remotely sensed landscape data, GIS-techniques, and simulation models offer promising tools for future large-scale research (see Section 6.4), but they must rely on empirical field studies at local scales. Clearly, a better understanding of the large-scale long-term impact of fragmentation on the landscape is required, yet the solution to the problems will more likely be found at a local scale. Richard T.T. Forman, a pioneer in landscape and road ecology at Harvard University, Massachusetts, put it simply: We must learn to 'think globally, plan regionally but act locally' (*sensu* Forman, 1995).