Landscape-scale Effects of Roads on Wildlife

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Via est vita
(AD 200)

Via vita est mortis
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ZUSAMMENFASSUNG

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Straßenverkehrsnetze haben zahlreiche Negativfolgen für wildlebende Tierarten: Ihnen droht der Verkehrstod durch Kollision mit Fahrzeugen, Straßen stellen eine Barriere für ihre Aktivitätsräume dar und beeinträchtigen Lebensräume durch vielfältige Störungen. Dessen ungeachtet nimmt das Straßennetz weltweit zu.

Die Qualität bisheriger straßenökologischer Forschung hat dieses Dilemma mit verursacht, denn: (1) Die untersuchten Forschungsfragen waren meist von wenig Relevanz für die praktische Straßenplanung. (2) Die gewählten Versuchsdesigns waren meist von geringer Qualität und erlaubten kaum abgesicherte Aussagen, aus denen sich brauchbare Empfehlungen für den praktischen Straßenbau ableiten ließen. (3) Viele Studien untersuchen die lokalen Effekte einzelner Straßen, während landschaftliche Wirkungszusammenhänge selten betrachtet wurden – ein Dilemma, da populationswirksame Effekte nur unter Berücksichtigung großräumiger Prozesse beurteilt werden können, und politische Entscheidungen meist auf der Landschaftsebene getroffen werden.

Die vorliegende Arbeit entwickelt eine Agenda für zukünftige straßenökologische Forschung. Zunächst werden fünf Forschungsfragen entwickelt, die von Relevanz für die praktische Straßenplanung sind. Für jede dieser Forschungsfragen werden hypothetische Versuchsdesigns entwickelt, die eine maximale Aussagefähigkeit ermöglichen.

Hiervon ausgehend liefert die vorliegende Arbeit einen Beitrag zur straßenökologischen Forschung auf Landschaftsebene. In einer landesweiten Analyse im Bundesland Hessen wird die Zunahme des Verkehrswegenetzes von 1930 bis 2002 dokumentiert. Das auf Ebene von Landkreisen und Naturräumen durchgeführte Monitoring beruht auf dem Index effektive Maschenweite, und ist eine Grundlage für jährliche Fortschreibungen. Es wird gezeigt, dass Verkehrsnetze negative Wirkungen auf Wildtierpopulationen haben. Die Bestände von Reh (*Capreolus capreolus*), Wildschwein (*Sus scrofa*) und Fuchs (*Vulpes vulpes*) sind kleiner je stärker die Zerschneidung eines Landkreises ist, gleichzeitig steigen die Zahlen der im Verkehr getöteten Tiere. Eine Untersuchung zum Feldhasen (*Lepus europaeus*) im schweizerischen Kanton Aargau zeigt ein ähnliches Bild. Je größer die Dichte stark befahrener Hauptstraßen desto kleiner die Populationsdichten. Vor diesem Hintergrund wird ein Modell entwickelt, das auf Grundlage von Straßen- und Landschaftsdaten Schwerpunkte von Kollisionen zwischen Wildtieren und Fahrzeugen voraussagen kann. Wildunfälle mit Reh und Wildschwein treten in den Dämmerungsstunden auf, insbesondere auf Straßen mit mittlerem Verkehrsaufkommen. Schwerpunkte sind Übergänge zwischen Wald- und Offenland, die beim Wechsel zwischen Schlaf- und Äsungsgebieten frequentiert werden. Die Modelle können dazu dienen Bestrebungen zur Vermeidung von Unfällen zu bündeln. Sie sind damit eine wichtige Grundlage eines Programms zur Entschneidung des bestehenden Verkehrswegenetzes.

Schlagworte: Verkehrsnetz, Straßenökologie, Forschungsagenda, Landschaftsebene, Monitoring, effektive Maschenweite, Wildunfälle, Verkehrsmortalität, Verkehrssicherheit

ABSTRACT

ROEDENBECK, I.A.: *Landscape-scale Effects of Roads on Wildlife.*
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Road networks affect wildlife in various ways. Animals are at risk of road mortality due to collisions with vehicles, roads act as a barrier for their seasonal and daily migrations, and cause various disturbance effects on wildlife habitat. This knowledge notwithstanding road networks continue to increase worldwide.

The dilemma was caused, among other reasons, by the quality of previous road ecology research, because: (1) the research questions addressed were not relevant for decision making. (2) The study designs used were of low quality preventing useful inferences for road construction. (3) Many studies focused on local-scale effects of a single road, neglecting landscape-scale interactions – a dilemma, because effects on population persistence can only be evaluated studying large-scale processes, and because the most pressing policy decisions are at the landscape scale.

The thesis in hand develops a research Agenda for future road ecology research. I present five research questions of direct relevance for road construction in practise. For any research question I present hypothetical study designs increasing inferential strength of the results.

Based on the Agenda, the thesis in hand contributes to the knowledge about landscape-scale road ecology research. A monitoring study in the federal state of Hesse, Germany, analyses the development of road networks from 1930 to 2002. Carried out on the level of administrative districts and natural areas the monitoring is based on index effective mesh size, and serves as a basis for periodic updates. I show that road networks affect wildlife populations. The abundance of roe deer (*Capreolus capreolus*), wild boar (*Sus scrofa*), fox (*Vulpes vulpes*), and badger (*Meles meles*) populations is small in administrative districts with a high degree of fragmentation, while at the same time number of road-kills increase. A study on brown hare (*Lepus europaeus*) in the Swiss Canton Aargau shows a similar pattern: The higher the density of heavily used roads, the smaller population abundances. Against this background I develop models predicting hotspots of vehicle-wildlife collisions on the basis of road and landscape data. Wildlife accidents with roe deer and wild boar take place in dawn, especially on roads with intermediate traffic densities. Hotspots are woodland-field interfaces frequented by animals when changing in-between resting places and pastures. Models are an essential basis for bundling mitigation efforts in the context of programs aiming at the de-fragmentation of the present road network.

Key words: road networks, road ecology, research agenda, landscape scale,
monitoring, effective mesh size, vehicle-wildlife collision, road moratlity, traffic safety

VORWORT

Der hier vorliegenden Text wurde von mir als Doktorarbeit an der Universität Giessen eingereicht. Die Arbeit basiert auf insgesamt fünf Manuskripten, die teilweise in wissenschaftlichen Fachzeitschriften veröffentlicht, und teils im Begutachtungsprozess zur Veröffentlichung befindlich sind. Alle Artikel sind in der veröffentlichten bzw. zu veröffentlichenden Originalsprache, zum Großteil also auf Englisch abgedruckt. Da zwei Arbeiten jedoch in deutschen Zeitschriften veröffentlicht sind, ist allen Arbeiten zum Verständnis sowohl eine deutsche, als auch eine englische Zusammenfassung vorangestellt. Der Abdruck der bereits veröffentlichten Artikel erfolgt unter Genehmigung der Herausgeberschaft der entsprechenden Zeitschrift.

I. Roedenbeck IA, Fahrig L, Findlay CS, Houlahan JE, Jaeger JAG, Klar N, Kramer-Schadt S, and van der Grift EA. 2007. The Rauischholzhausen-Agenda for Road Ecology. *Ecology and Society* 12(1): 11. [online] URL: http://www.ecologyandsociety.org/vol12/iss1/art11/

II. Roedenbeck IA, Esswein H, and Köhler W. 2005. Landschaftszerschneidung in Hessen – Entwicklung, Vergleich zu Baden-Württemberg und Trendanalyse als Grundlage für ein landesweites Monitoring (mit Kartenbeilage). *Naturschutz und Landschaftsplanung* 37(19): 193-300.

III. Roedenbeck IA, and Köhler W. 2006: Effekte der Landschaftszerschneidung auf die Unfallhäufigkeit und Bestandsdichte von Wildtierpopulationen - Zur Indikationsqualität der effektiven Maschenweite. *Naturschutz und Landschaftsplanung* 38 (10/11): 314-322.

IV. Roedenbeck IA, and Voser P. 2007. Effects of roads on spatial distribution, abundance and road mortality of brown hare (*Lepus europaeus*) in Switzerland. *European Journal of Wildlife Research* (in review)

V. Roedenbeck IA, and Köhler W. 2007. Temporal and spatial characteristics of vehicle-wildlife accidents in Germany. *Journal of Applied Ecology* (in review)

Die Manuskriptform wurde beibehalten, um dem Leser einen leichteren Zugang zur Thematik zu ermöglichen, denn alle fünf Studien basieren auf unterschiedlichen Datensätzen. Obgleich die Artikel inhaltlich aufeinander aufbauen, bearbeiten sie klar voneinander abzugrenzende Forschungsfragen. Somit findet sich in jedem Kapitel eine Einleitung in die spezifische Forschungsfrage, relevante Methoden, Ergebnisse, sowie eine ergebnisspezifische Diskussion mit Literaturverweisen. Die fünf Artikel werden von einer generellen Einleitung und Diskussion eingerahmt (Kapitel 1 und 7), welche die Artikel generell und umfassend im Forschungsfeld einordnen.

Die vorliegende Arbeit wurde von einem Promotionsstipendium der Deutschen Bundesstiftung Umwelt gefördert, die einen klaren Schwerpunkt auf praxisorientierte Arbeiten legt. Ich werde in der abschließenden Diskussion deswegen besonders die Praxisrelevanz jeder einzelnen Studie im Rahmen politischer Entscheidungsprozesse beleuchten.

Inga A. Roedenbeck

PREAMBLE

The study in hand has been submitted as doctoral thesis at University of Giessen. The thesis is based on five manuscripts, in parts being published in scientific journals, and in parts being in the review process for publication. All articles are printed in the original language submitted to the journals. Hence, this thesis is in English in large parts. However, as two articles have been published in German journals, I will provide a German and an English abstract for each article. All published papers are reproduced with permission from the publishers.

I. Roedenbeck IA, Fahrig L, Findlay CS, Houlahan JE, Jaeger JAG, Klar N, Kramer-Schadt S, and van der Grift EA. 2007. The Rauischholzhausen-Agenda for Road Ecology. *Ecology and Society* 12(1): 11. [online] URL: http://www.ecologyandsociety.org/vol12/iss1/art11/

II. Roedenbeck IA, Esswein H, and Köhler W. 2005. Landschaftszerschneidung in Hessen – Entwicklung, Vergleich zu Baden-Württemberg und Trendanalyse als Grundlage für ein landesweites Monitoring (mit Kartenbeilage). *Naturschutz und Landschaftsplanung* 37(19): 193-300.

III. Roedenbeck IA, and Köhler W. 2006: Effekte der Landschaftszerschneidung auf die Unfallhäufigkeit und Bestandsdichte von Wildtierpopulationen - Zur Indikationsqualität der effektiven Maschenweite. *Naturschutz und Landschaftsplanung* 38 (10/11): 314-322.

IV. Roedenbeck IA, and Voser P. 2007. Effects of roads on spatial distribution, abundance and road mortality of brown hare (*Lepus europaeus*) in Switzerland. *European Journal of Wildlife Research* (in review)

V. Roedenbeck IA, and Köhler W. 2007. Temporal and spatial characteristics of vehicle-wildlife accidents in Germany. *Journal of Applied Ecology* (in review)

I retained the manuscript form to enable an easy access to the topic, because each article is based on different base data. Though articles are sequenced, they analyse separate research questions. Consequently, each chapter consist of an introduction into the specific research question, relevant methods, results, and a specific discussion including references. The five articles are framed by a general introduction and discussion (chapter 1 and 7), discussing the overall importance of the results against the background of the current knowledge in road ecology research.

The thesis in hand was funded by a scholarship of the German Environmental foundation, which places emphasis on practice orientated research. In the concluding discussion I will consequently focus on the pratical relevance of each paper in the context of political decision making.

Inga A. Roedenbeck

CONTENT

Chapter 1.
Landscape-scale Effects of Roads on Wildlife

A General Introduction

Road ecology is a comparatively recent field of research, which developed in the latest decades when the ancient grown road network multiplied enormously. Modern roads reach a much wider land area than the physical occupation of land by the road per se. It is estimated that transportation infrastructure affects at least 19% of the United States conterminous land area (Forman 2000), and 20% of the Netherlands (Reijnen et al. 1995). Thus, numerous field studies, conference proceedings, reviews and reports have been published about the effects of roads on wildlife (e.g. Canters et al. 1997, Evink et al. 1998, 1999, 2001, Forman et al. 2002, Glitzner et al. 1999, Holzgang et al. 2000, Irwin et al. 2003, Iuell et al. 2003, Trocmé et al. 2003). It is conspicuous that many studies focus on the local effects of a single road, while just a few analyse landscape-scale effects. However, the landscape-scale perspective is important, because population persistence is regulated by processes operating at larger scales, and the most pressing policy and management issues are generally not at the local, but at the landscape scales (National Research Council 2005).

The thesis in hand intends to contribute to the knowledge about landscape-scale effects of roads on wildlife populations. I will start with a sight into past, because the current road network has developed on the basis of ancient conditions and necessities. They should be considered, as they are still used as arguments for an economic and socially driven claim for a further development of road networks. Afterwards, I will give a general idea about the effects of roads on wildlife. I will summarize most important findings, and present a structuring of the primary road effects. Finally, I will discuss why the landscape-scale perspective is crucial in road ecology, and conclude with the research questions this thesis intends to answer.

1.1 HISTORICAL DEVELOPMENT OF ROAD NETWORKS

1.1.1 ANCIENT PATHS, ROMAN ROADS AND BAROQUE AVENUES

The first ´roads´ used by humans were probably tracks made by animals, which people adopted for transport (Seiler 2003b). Due to the invention of wheels, enabling the transport of goods on wagons, paths were developed. Early records of wagons date back to before 4000 BC from different regions: the Northern Caucasus, Mesopotamia and the Indus culture Harappa. First chariots have been developed around 2500 BC and were used by Hittites, Assyrians and Egyptians. In Europe, the history of natural paths dates back to the Germanic and Celtic prehistory. Such ´natural paths´ (´*Naturwege*`) were unfortified and did not demand structural measures. As there were no frontiers and boundaries in the sparsely populated country, everybody could choose the most convenient path without asking for permission. The route of such paths followed geology and topography of the terrain (**Fig. 1.1**). Routes alongside ridges were preferred, because valleys were often covered with floodplain forests, and meandering rivers and estuaries were insuperable barriers for wayfarers. Also, dangers and hazards were visible from a distance. Ancient paths in Europe mainly served for trade and transportation of goods to fortified villages.

Actual ´roads´ were developed during the Roman Empire (*via strata*) in Europe. Roman roads marked a novelty in this age. With fortifications at important positions in terrain they first and foremost served for military purposes. In contrast to the ancient paths Roman roads were passable irrespective of soil moisture, and paved straight ways through forests, plains and even mountains (**Fig. 1.1**). These roads demanded maintenance and construction costs procured by the State of Rome, the Roman provinces, municipalities or the particular landholders, respectively, depending on road type (*via publica, via militaris, via vicinalis, via privata, via urbica*) (Adam 1994). The technical secret of success was the roads´ layer composition (Chevallier 1976). Clearings and excavations of about one meter depth were necessary and essential to fix the ground. Roads were laid upon a well-constructed embankment of varying height, in order to give them a properly drained base (Margary 1973). Stones, gravel and sand layers were displayed until the road surface was fixed with cobblestones.

Fig. 1.1. *Historic development of roads in Europe: Ancient path in Hesse (left), Roman road ´Via Appia` (centre), and modern highways in Hesse (right) (photos by: Mechelhoff, Wikimedia, and Fritz 2005).*

After the collapse of the Roman Empire (about 455 AD) the Merovingian included Roman roads and old ancient paths into their road network. Monasteries were built up at important junctions, available paths became pilgrims´ ways and trade roads were used for goods traffic to villages and marketplaces. Colonization took place alongside these roads, castles and monasteries became starting points for settlements. As various small states existed in Central Europe during medieval times, monarchs did not see any necessity for investing in fortified roads. However, with growing valuation of road networks´ contribution to economy and welfare, planning, construction and financing of roads was covered by the State since in the middle of the 19th century, when artificial avenues (Chaussee) were constructed. Old paths following ridges were abandoned, and roads started following valleys and rivers. Avenues often consisted of a stone road, and a summer path. The stone road was the paved part with a base layer of gravel and a surface of sand and adobe. The summer path for unshod animals was adjacent to the stone road, unpaved and not usable in winter. Baroque avenues were the starting point for the European country roads we know today.

1.1.2 Modern Roads

The construction of modern roads was initialized by the invention of the automobile in 1886 in Germany. Until the 18th century coaches and horses had been the only means of transportation. Railroads increased travel speed, however, people were bounded to timetables and defined stopping places. The automobile changed mobility behaviour fundamentally. It enabled the universal und individual mobility of people as well as a flexible and fast transport of heavy goods. Consequently, there was an increasing demand for straight high-speed routes and networking of transportation ways, resulting in the construction of multilane highways and expressways, and a densification in road networks in the 20th century (Lay 1994).

There have been various reasons for constructing roads in the course of time: They granted access to food and accommodation, they served as routes for seasonal migrations, for pilgrimage and trade. Military and national-political considerations have been common reasons for road construction. However, in the course of modern times roads started to fulfil further social and economical needs, the access to work, education and amusement. Caused by the development of social ideals, culminating in boundless mobility and globalization, the road network increased enormously in the latest decades. Between 1970 and 1996, the length of the Trans-European Transport Network (TEN-T, including highways carrying long distance traffic) almost doubled, to cover 1.2% of the available land area. Today, the network is made up of ca. 75,000 km of roads (ca. 20,500 km of which are being planned) (Iuell et al. 2003). This trend of a steadily increasing road network is very likely to continue in most parts of the world (e.g. NRTF 1997).

Following a road history of more than 2000 years there was relatively little regard for ecology until the recent years. In the mid-nineteenth century, Henry David Thoreau first described the results of a direct wagon-wheel hit on a turtle (cited in Forman et al. 2002), and in 1925 Dayton Stoner raised concern about traffic-killed vertebrates he observed on a summer drive (Stoner 1925). Within a few decades the road network reached such a dimension that ecological affects became highly visible.

1.2 EFFECTS OF ROADS ON WILDLIFE

Effects of modern roads on wildlife have been structured and ordered by different authors in various ways (e.g. Forman and Alexander 1998, Forman et al. 2002, Jackson 2000, Spellerberg 1998, Trombulak and Frissell 2000, Underhill and Angold 2000, Seiler 2003a) (**Tab. 1.1**). Up to now there is no consistent classification, as structuring is dependent on the perspective of interest, the scale, and the taxa under investigation. In contrast to previous classifications I distinguish between three primary road effects focussing on wildlife and its basic requirements: (1) effects on habitat, (2) effects on movement, and (3) mortality. Effects on animals´ behaviour are the basic process underlying different factors, so I do not refer to behaviour as a separate point. The three primary road effects are interdependent and may act contradictory or cumulative. Some effects mesh, like for example barrier effects and road mortality. If animals are hindered from road crossing because they are killed on the road, this leads to the same barrier effect as if animals avoid road crossings. As another example, habitat degradation by noise and light leads to road avoidance causing barrier effects. This interdependence of factors makes structuring difficult and allows several solutions for grading (**Tab. 1.1**).

In addition to the ´primary´ effects, roads also cause various ´secondary´ effects. They increase the access to wildlife areas to hunters, poachers and tourists (Gratson and Whitman 2000), and new roads into forested landscapes lead to deforestation and further economic developments (Chomitz and Gray 1996). In this chapter (´Effects of roads on wildlife´), I will first and foremost focus on the processes underlying primary local-scale effects, and refer to the landscape-scale effects and their underlying processes in the following chapter (´The landscape scale´).

***Tab. 1.1**. Primary ecological effects of roads on wildlife. A comparison of different classifications is shown. Numbers indicate structuring in the text.*

Roedenbeck (2007)	Forman et al. (2002)	Iuell et al. (2003)	Jackson (2000)	Forman & Alexander (1998)	Trombulak & Frissell (2000)	Underhill & Angold (2000)
(1) Effects on habitat						
(1.1) Effects on habitat amount Direct habitat loss due to physical occupation.	**Changes in habitat amount** Habitat loss	**Habitat loss** Direct habitat loss and indirect effects of disturbance.	**Direct habitat loss**			
(1.2) Effects on habitat quality Reduced habitat quality due to pollution and disturbance	**Changes in habitat quality** reduced or improved habitat quality	**Disturbance/ Edge Effects**	**Degradation of habitat quality**	**Disturbance**		**Disturbance Effects**
(1.2.1) **Pollution of the biological environment** Disturbance due to traffic noise, light, and increased human access		Pollution of biological environment (traffic noise and light).	**Increased human exploitation**		**Increased human access** Hunting, fishing, recreation, and changes in land use.	Wind, increased human noise and access.
(1.2.2) **Pollution of the chemical environment** Disturbance due to dust, salt, heavy metal, nutrients, toxins.		Pollution of chemical environment (dust, salt, heavy metals, nutrients, toxins).			**Alteration of the chemical environment** Salt, organic molecules, ozone, heavy metals, nutrients	**Pollution** dust, salt, exhaust (nitrogen oxides, ozone, organic gases, heavy metals)
(1.2.3) **Pollution of the physical environment.** Disturbance causing effects on soil, runoff, surface-water flow, sedimentation.		Pollution of physical environment (construction).		**Effects on water and sediment**	**Alteration of the physical environment** Effects on soil, surface-water flow, runoff, sedimentation	**Changes in local hydrology** Increased runoff, changes in streamflow
(2) Effects on movement						
(2.1) Barrier effects Decreased movement alongside or across roads.	**Effects on landscape connectivity** Barrier effects	**Barrier Effects**	**Disruption of social structure** by sex-dependent permeability	**Barrier Effects**		**Physical barriers to the movement**
(2.1.1) **Road avoidance** Due to disturbance effects, and reduced habitat quality.			**Road avoidance**	**Road avoidance**		Physical arrier effect
(2.1.2) **No road crossings** Due to road mortality.						Road mortality
(2.2) Corridor effects Increased movement alongside roads (positive or negative).	increased movement	**Corridor habitats** (Positive and negative)			**Spread of exotic species**	**Provision of habitat and corridors**
(3) Mortality						
Mortality Due to collisions with cars.	**Mortality**	**Mortality**	**Mortality**	**Mortality**	**Mortality from collision with vehicles** **Mortality from road construction**	Road mortality is a consequence of physical barrier effects

1.2.1 EFFECTS ON HABITAT

Roads affect wildlife habitat in two different ways. First, the physical occupation of land caused by road construction leads to a direct habitat loss (**Tab. 1.1**, 1.1). Second, maintenance and use of road cause various effects altering habitat quality (**Tab. 1.1**, 1.2).

Most habitat alterations are negative for wildlife resulting from pollution of the biological (**Tab. 1.1**, 1.2.1), chemical (1.2.2) and physical environment (1.2.3) (Seiler 2003a). The main biological factor is disturbance caused by traffic noise, light and the presence of vehicles. Especially birds are affected by traffic noise, as it directly interferes with their vocal communication and thereby affects their territorial behaviour and mating success (Illner 1992, Reijnen et al. 1995, 1996). For example, lapwing (*Vanellus vanellus*) and black-tailed godwit (*Limosa limosa*) are disturbed near roads (Van der Zande et al. 1980). Such disturbances cause modifications of animal´s behaviour and alterations in physiological state. For example, northern spotted owls (*Strix occidentylis caurina*) living close to a forest road experience higher levels of a stress-induced hormone than individuals living in areas without roads (Wasser et al. 1997). Also, female big-horn sheep heart rate and metabolic rate is known to increase near a road (MacArthur et al. 1979).

The chemical disturbance of habitats is caused by five different general classes of pollutants: heavy metals, salt, organic molecules, ozone, and nutrients (Farmer 1993, Muskett and Jones 1980, Thompson and Rutter 1986, Trombulak and Frissell 2000). The physical disturbance mainly refers to soil density, temperature, surface-water flow, run-off patterns, and sedimentation (Maltby et al. 1995). While biological disturbances show direct effects on animal behaviour, chemical and physical factors indirectly affect wildlife populations by degradation of habitat and natural resources. This leads to a decrease in habitat quality or indirect habitat loss.

Although most effects of roads on habitat quality are negative, some species respond positively to habitat near roads (Forman et al. 2002). For example, some road verges are inhabited by disturbance-tolerant species, and verge habitat following roads may contain relatively high species richness in comparison to adjacent fields (Adams and Geis 1983, Laursen 1981, Port and Thompson 1980, Roach and Kirkpatrick 1985).

1.2.2 EFFECTS ON MOVEMENT

Effects of roads on animal movement may as well be positive or negative. On the one hand, roads and traffic may act as barriers decreasing movement rates (**Tab. 1.1**, 2.1). On the other hand road verges can act as corridors increasing movement alongside roads (**Tab. 1.1**, 2.2).

The barrier effect of roads and traffic is caused by both: road avoidance (Tab. 1.1, 2.1.1), and abortive crossings due to road mortality (Tab. 1.1, 2.1.2; see chapter 1.2.3 below). Road avoidance is caused by the physical presence of a road and/or habitat disturbance leading to modifications in animal behaviour such as altered escape responses. For example, it has been shown that roads act as barriers for the movement of small animals, such as carabid beetles, forest-dwelling mice (Mader 1984), and white-footed mice (Peromyscus leucopus) (Merriam et al. 1989). The frequency of road-crossings by medium-sized animals, e.g. brown-hare (*Lepus europaeus*), grey squirrel (*Sciurus carolinensis*), and stoat (*Mustela erminea*) is reduced with increasing road width (Oxley et al. 1974). And also large mammals, such as mule deer (*Odocoileus hemionus*), elk (*Cervus canadensis*), Roosevelt elk cows during calving (*Cervus elaphus roosevelti*), and wolves (*Canis lupus*) avoid the proximity to heavily used roads (Rost and Bailey 1979, Thurber et al. 1994, Witmer and deCalesta 1985). Grizzly bears (*Ursus arctos*) were found to strongly avoid areas within 500 m of a highway (Waller and Servheen 2005), and black bears (*Ursus americanus*) prefer crossing roads during low traffic volumes, and cross low-traffic-volume roads relatively more frequently than high-traffic-volume roads (Brody and Pelton 1989).

Increased movement of individuals in road verges aids the dispersal of several plants and animal species. However, especially stress-tolerant invasive species benefit (Brothers and Spingarn 1992, Hess 1994, Macdonald and Frame 1988), e.g. the introduced cane toad (*Bufo marinus*) in Australia (Seabrook and Dettmann 1996). As these species pre-dominate native populations, negative competition effects by far outweigh positive dispersal effects. Furthermore, there is a risk attached to corridors. If they fail to provide a throughway to favourable habitat at a reachable distance they may function as a sink habitat whilst at the same time depleting the source population (Pulliam 1988, Vermeulen 1994). Wolves (*Canis lupus*) seem to be attracted by some roads as they provide easy travel corridors and greater access to prey. However, these are seasonally closed roads with limited human use while roads year-round open to public use are avoided (Thurber et al. 1994).

1.2.3 MORTALITY

Direct mortality results from collisions of wildlife with vehicles on the road (**Fig. 1.2**). For endagered species in small (sub-) populations traffic mortality may be a direct threat to population viability (Forman and Alexander 1998). For example, collisions with vehicles accounted for 49% of mortality of the endangered Florida panther (*Puma concolor coryi*) (Maehr et al. 1991), and the population of only about twenty individuals is unlikely to be able to sustain this pressure. Road kills are a significant cause of mortality for the endangered ocelot (*Leopardus pardalis*) (Hewitt et al. 1998), the Iberian lynx (*Lynx pardinus*) in Spain (Ferreras et al. 1992), and wolves (*Canis lupus*) in Minnesota (Fuller 1989). Road kill is also the largest mortality source for moose (Alces alces) in the Kenai National Wildlife Refuge in Alaska (Bangs et al. 1989), and for barn owls (*Tyto alba*) in the United Kingdom (Newton et al. 1991). Amphibians are especially sensitive to road mortality, as their seasonal migration to breeding sites often leads them across a road (Van Gelder 1973). The likelihood of road-kills for amphibians was calculated to be 34-61% on a low-traffic country road, and up to 98% for highly used roads such as motorways (Hels and Buchwald 2001). Hence, many roads may be more or less impassable for amphibians subject to their traffic volume and width (Kostrzewa 2006).

Road kill is a classical death-trap phenomenon. Animals are attracted to roads for a variety of reasons and often become road-kills themselves. For example, snakes and amphibians are attracted by the warm asphalt, mammals search for de-icing salt, and some carnivores find roads to be efficient travel ways.

Traffic mortality is a growing problem not only for conservation purposes, but also for wildlife management, traffic safety and human health (Groot Bruinderink and Hazebroek 1996). Collisions with large mammals such as roe deer, wild boar and moose cause substantial material damages and injuries. As a consequence, traffic safety is the driving force behind mitigation efforts against fauna casualties (Seiler 2003a).

Fig. 1.2*. Attempts to cross a road often result in wildlife causalities. Common toad (Bufo bufo, left) and hedgehog (Erinaceus europaeus, centre) are susceptible for traffic mortality in Germany (photos: Roedenbeck 2006). Collisions between vehicles and large mammals such as wild boar (Sus scrofa, right) are a serious problem for traffic safety (photo: Polizeipräsidium Westhessen 2006).*

1.3 THE LANDSCAPE SCALE

A goal of ecological research is to understand how the environment affects the abundance and distribution of organisms, while processes considered are typically at a "local" scale, i.e., at the same scale or smaller than the scale of the abundance/distribution pattern of interest (Fahrig 2003). Landscape ecology, as a subdiscipline of ecology, surveys the effect of pattern on process (Turner 1989), and studies how landscape structure affects the processes that determine the abundance and distribution of organisms. Landscape ecology addresses the causes and consequences of spatial heterogeneity (Forman 1995). Spatial heterogeneity is based on the amount of entities (e.g. habitat types) and their spatial arrangement (Pickett & Cadenasso 1995), which both influence the connectivity of a landscape. Landscape connectivity is the degree to which the structure of a landscape helps or hinders the movement of wildlife species (Taylor et al. 1993).

1.3.1 IMPORTANCE OF THE LANDSCAPE PERSPECTIVE IN ROAD ECOLOGY

A newly constructed road always adds to an existing road network. As a consequence, it does not only affect the local habitat patch, but the amount and spatial arrangement of habitat patches in the entire landscape. By alteration of habitat amount and spatial arrangement, road networks affect the structure and spatial heterogeneity of a landscape. When habitat is lost, remaining habitat patches become smaller and more isolated from each other (Fahrig 2003a). The increasing distance between available habitat patches, the additive barrier and mortality effects of roads, limit dispersal possibilities of species and decrease landscape connectivity. Loss of landscape connectivity is landscape fragmentation, which is suspected to be one of the most important factors causing population declines (Forman et al. 2002). Assumed consequences for wildlife are first of all a disruption of horizontal processes maintaining regional populations (Forman 1998), such as blocking of dispersal and re-colonization, inaccessibility of resources and mates, and hindered individual exchange. Possible consequences are decreasing population sizes, isolation of populations, reduced genetic variability with altered reproductive success, inbreeding and reduced fitness (Baker 1998). The overall consequences are decreased long-term population persistence and loss of biodiversity (Forman & Alexander 1998) (**Fig. 1.3**).

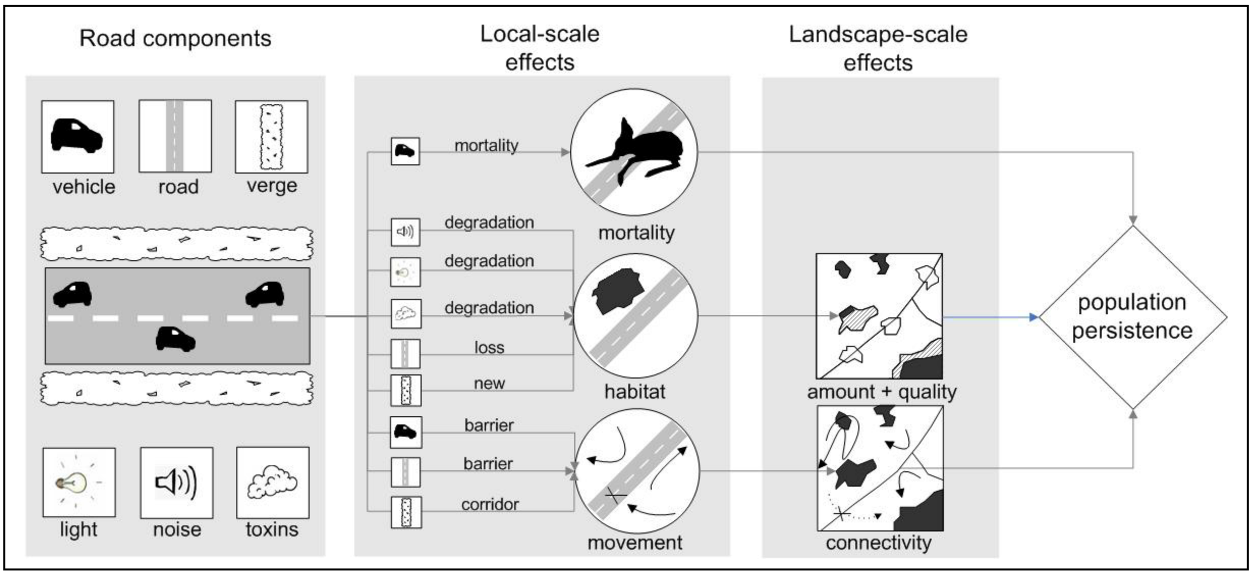

Fig. 1.3. *Effects of roads on population persistence at the local and the landscape scale.*

1.3.2 PROBLEMS OF THE LANDSCAPE PERSPECTIVE IN ROAD ECOLOGY

Surprisingly, a lot of studies analyzing the effects of landscape fragmentation, in general, are conducted at the local scale of individual patches and not landscapes (Fahrig 2003b), though landscape fragmentation is a landscape-scale process. And in road ecology, in particular, there are numerous studies describing local-scale road effects referring to local habitat, local movements and local mortality. Hence, though landscape-scale effects of roads on population persistence are assumed, the mechanisms described above are not proven yet.

The underlying problem of landscape-scale studies is that road effects show a lagged response because the different effects of roads on wildlife populations typically occur at different rates (Forman et al. 2002). Habitat loss is the most immediate effect. However, the increase in population mortality due to wildlife-vehicle collisions takes a little longer to be observable, and the effect of roads as barriers, reducing landscape connectivity, will likely take several generations to be observed (Findlay and Bourdages 1999). Consequently, landscape-scale studies often require complex long-term experimental designs and large-scale data. An additive problem is that ecological complexity, especially at large scales, tends to blur research results (Bissonette and Storch 2002), and this often results in researchers designing less complex local-scale studies and trying to draw conclusions about landscape-scale effects. However, local results can not easily be extrapolated to infer effects on population persistence. Issues of extrapolation mainly result from two reasons: (1) road mortality is regulated by population dynamics and may be compensatory; and (2) local movement patterns and habitat loss may be altered by landscape-scale population and meta-population dynamics.

(1) The mortality problem: Many local studies counted road kill numbers on single streets (e.g. Kuhn 1987, Kutzer and Frey 1979, Lodé 2000), and it has also been shown that road kill rates of some species, e.g. otters and moose (Hauer et al. 2002, Seiler 2004), increased over time, possibly due to increased traffic densities. Since mortality depends on population abundance, it is hardly interpretable without information on population sizes, and the likelihood of compensatory mechanisms such as reduction in other mortality resources or increase in reproduction in response to road mortality. None of the studies carried out thus far provides such information and was carried on long enough to detect long-term compensatory mechanisms. Consequently, large numbers of casualties may not necessarily imply a threat to the survival of a species, but rather indicate that it is abundant and widespread (Seiler 2003b). For some species, such as badger and butterflies, it has already been demonstrated that road mortality does not affect population persistence (Reicholf 1983, Munguira and Thomas 1992). Even though some local hedgehog populations seem to be affected by road traffic (Huijser et al. 1998), overall population persistence does not seem to be in decline (Eichstädt and Roth 1997). *Summa summarum*, road kills do not seem to have detrimental effects on animal populations except in those cases of species with small or diminishing populations (Forman and Alexander 1998, Spellerberg 1998).

(2) The landscape-scale problem: Processes operating at the landscape-scale may alter or inhibit local-scale patterns with the consequence that population persistence is not affected. Looking at local disturbance, for example, one might argue that populations in response to disturbance alter their distribution, without growing smaller (Van der Zande 1980). Black bears (*Ursus americanus*), for instance, react to increases in road densities by shifting the locations of their home ranges to areas of lower road densities

(Brody and Pelton 1989). Also, the reduction of grey partridge (*Perdix perdix*) density near roads may be compensated by an increase in density away from roads (Illner 1992). It is also possible, that barrier effects taking place at a local scale, do not affect population persistence due to metapopulation dynamics at the landscape scale. Subpopulations can stay alive, and extinct subpopulations can be re-colonized, as long as a definite exchange of individuals and gene flow is granted. Whether and how fast such re-colonizations are successful, depends on the permeability of existing barriers and the degree of landscape connectivity or landscape fragmentation, respectively. Focal questions refer to thresholds, for example: what is the critical road density in an area, above which a population cannot persist?

1.3.3 Current state of Knowledge about landscape-scale effects

As a consequence of the problems described above concerning methodology, extrapolation and interpretation, little is known about the landscape-scale effects of roads thus far. Most findings refer to amphibians, because they accumulate at few locations twice a year, enabling easy counting of population size and characterizing landscape patterns in nested buffers around ponds (Houlahan et al. 2006). With the help of such nested buffer designs it has been shown that traffic density is negatively correlated with pond-occupation probability of moor frogs (*Rana arvalis*) (Vos and Chardon 1998), abundance of roadside anuran populations (Fahrig et al. 1995), and leopard frog (*Rana pipens*) population abundance (Carr and Fahrig 2001). Species richness of herptiles was negatively correlated with the density of paved roads on lands up to 2km from wetland (Findlay and Houlahan 1997). There are also few amphibian studies showing genetic effects of roads, for example, a reduced genetic variation in common frog (*Rana temporaria*) has been demonstrated (Reh and Seiz 1990, Johansson et al. 2005). It has been shown that genetic distances of common frog breeding sites in the City of Brighton, with a mean geographic distance of 2.3 km, were almost twice as high as those between rural sites with a mean distance of 41 km (Hitchings and Beebee 1997).

Few studies report landscape-scale effects on mammal species. One study analyzed dispersal patterns of roe deer (*Capreolus capreolus*) showing a negative correlation between the density of barriers in a landscape and the distance travelled from birth to death location. In the same study, in areas with low barrier density more roe deer individuals got older than two years (Müri 1999). In a landscape-scale study in The Netherlands, road density was the most important variable related to the decline in badger (*Meles meles*) setts, because abandoned setts were situated in areas with the highest road density and the highest increase in road density in the last 20 years (van der Zee et al. 1992). There is also one study showing genetic subdivision due to roads for a mammal species, the bank vole (*Clethrionomus glareoulus*) (Gerlach and Musolf 2000).

Very few landscape-scale studies report a road density threshold necessary for sustainable mammal populations. Wolves (*Canis lupus*) in Michigan and Ontario do not occupy areas with road densities beyond 0.58 km/km² (Mech et al. 1988), and similar road density thresholds were reported for brown bears (*Ursus arctos*) (Clevenger et al.

1997). However, the wolve study (Mech et al. 1988) analysed a very small sample size of 9 landscapes, and did not investigate any other potential impact factors. Hence, these studies only scratch the surface of the research problem. Against the background of the limited knowledge, I conclude that further results about landscape-scale road effects are urgently required.

1.4 AIMS OF THE STUDY IN HAND

The study in hand shall contribute to the knowledge about landscape-scale effects of roads on wildlife populations. The study is not just basic research, but should be an active contribution to political decision making. The thesis is based on five analyses presented in chapters 2, 3, 4, 5 and 6.

Chapter 2 introduces the problem of road ecology research, figuring out that former road ecology studies did (and still do) not have any influence on decision making. The chapter discusses two reasons for this, arguing that the research questions asked were most often not relevant for the decision making process, and that a multitude of studies was conducted using low-quality study designs decreasing inferential strength of research results. Based on these fundamental ideas we show options for action to increase the influence of future research in transportation planning. We do this by developing five questions of high practical relevance for the decision making process, and developing study designs of high inferential strength. The question on ´landscape-scale road effects´ is one of the five crucial questions outlined for future road ecology research.

Chapter 3 illustrates the historical development of road networks in the federal state of Hesse in Germany from 1930 to 2002. This paper introduces the environmental problem of landscape fragmentation, and serves as a basis for further analysis. To increase the impact on decision making I apply a fragmentation index being well known in the political area in Germany.

Chapter 4 proceeds with the crucial linkage between pattern and process, investigating the effects of landscape fragmentation on wildlife populations. The study is conducted in Hesse on the same spatial level as the historical analysis, and uses hunting statistics for roe deer (*Capreolus capreolus*), wild boar (*Sus scrofa*), fox (*Vulpes vulpes*), and badger (*Meles meles*) as base data for population abundance. We ask, whether or not there is any relationship between road network density and the abundance of wildlife populations.

Chapter 5 poses the same research question as chapter 4, but changes the study area from Hesse to Canton Aargau in Switzerland, and the species of interest to brown hare (*Lepus europaeus*) populations. Again we ask, whether or not there is any relationship between road network density and the abundance of wildlife populations. However, we manage to improve study design and inferential strength with the help of high quality wildlife data and a species of conservation concern.

Chapter 6 investigates locations of vehicle-wildlife accident sites in a state wide analysis in Hesse. I develop landscape-scale models predicting accident hotspots based on landscape characteristics. This concluding analysis is a contribution to troubleshooting, aiming at recommending mitigation measures at collision spots, and thereby mitigating vehicle-wildlife accidents. The predictive models developed may serve as an essential basis for a state wide de-fragmentation program in Germany.

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Chapter 2.
The Rauischholzhausen-Agenda for Road Ecology

Increasing the influence of road research in transportation planning

Roedenbeck IA, Fahrig L, Findlay CS, Houlahan JE, Jaeger JAG, Klar N, Kramer-Schadt S, and Van der Grift EA. 2007. The Rauischholzhausen agenda for road ecology. *Ecology and Society* 12(1): 11. [online] URL: http://www.ecologyandsociety.org/vol12/iss1/art11/

Abstract

Despite the documented negative effects of roads on wildlife, ecological research on road effects has had comparatively little influence on road planning decisions. We argue that road research would have a larger impact if researchers carefully considered the relevance of the research questions addressed and the inferential strength of the studies undertaken. At a workshop at the German castle of Rauischholzhausen we identified five particularly relevant questions, which we suggest provide the framework for a research agenda for road ecology: (1) Under what circumstances do roads affect population persistence? (2) What is the relative importance of road effects vs. other effects on population persistence? (3) Under what circumstances can road effects be mitigated? (4) What is the relative importance of the different mechanisms by which roads affect population persistence? (5) Under what circumstances do road networks affect population persistence at the landscape scale? We recommend experimental designs that maximize inferential strength, given existing constraints, and we provide hypothetical examples of such experiments for each of the five research questions. In general, manipulative experiments have higher inferential strength than do nonmanipulative experiments, and full before-after-control-impact designs are preferable to before-after or control-impact designs. Finally, we argue that both scientists and planners must be aware of the limits to inferential strength that exist for a given research question in a given situation. In particular, when the maximum inferential strength of any feasible design is low, decision makers must not demand stronger evidence before incorporating research results into the planning process, even though the level of uncertainty may be high.

Zusammenfassung

Die Rauischholzhausen-Agenda zur Straßenökologie. *Obwohl die negativen Effekte von Straßen auf Wildtiere zunehmend dokumentiert werden, hat die straßenökologische Forschung bislang vergleichsweise wenig Einfluss auf Planungsentscheidungen im Straßenbau. Wir argumentieren, dass die Straßenökologie ihren Einfluss stärken könnte, wenn mehr Sorgfalt bezüglich der Relevanz der ausgewählten Forschungsfrage sowie der statistischen Aussagekraft der Untersuchungen an den Tag gelegt werden würde. Im Rahmen eines Workshops auf Schloss ´Rauischholzhausen´ wurden fünf besonders relevante Forschungsfragen identifiziert, die wir als Leitfaden einer zukünftigen Forschungsagenda in der Straßenökologie vorschlagen: (1) Unter welchen Bedingungen beeinflussen Straßen die Überlebensfähigkeit von Populationen? (2) Was ist die relative Bedeutung von Straßeneffekten im Verhältnis zu anderen potentiellen Effekten auf die Überlebensfähigkeit von Populationen? (3) Unter welchen Bedingungen können Straßeneffekte verhindert werden? (4) Welche relative Bedeutung haben ver-schiedene Straßeneffekte für die Überlebensfähigkeit von Populationen? (5) Unter welchen Bedingungen beeinflussen Straßenverkehrsnetze die Überlebens-fähigkeit von Populationen auf derLandschafts-ebene? Wir entwickeln Versuchsdesigns, die unter Anerkennung bestehender Rahmenbedingungen, eine maximale Aussagekraft ermöglichen. Darüber hinaus werden, für jede der fünf Forschungsfragen, hypo-thetische Beispiele der vorgeschlagenen Versuchs-designs besprochen. Grundsätzlich haben manipulative Experimente eine höhere Aussagekraft als nicht-manipulative Experimente, und vollständige ´before-after-control-impact´ Designs sind ´before-after´ oder ´control-impact´ Designs vorzuziehen. Sowohl Wissenschaftler, als auch Planer müssen sich stets der Einschränkungen bewusst sein, die für die Aussagekraft einer bestimmten Forschungsfrage in einer gegebenen Situation bestehen. Insbesondere, wenn die maximale Aussagekraft jedes potentiell durchführbaren Versuchsdesigns gering ist, dürfen Entscheidungsträger keine erhöhte Beweiskraft verlangen bevor sie Forschungsergebnisse im Planungsprozess berücksichtigen – selbst wenn mit dem vorgelegten Ergebnis eine hohe Unsicherheit assoziiert ist.*

2.1 INTRODUCTION

Mobility of people and goods is an essential component of the modern world, with its emphasis on globalization and economic opportunity. However, the transportation infrastructure that enhances connectivity among human settlements often results in decreased connectivity among remaining natural habitats and wildlife populations. It is estimated that the transportation infrastructure affects at least 19% of the conterminous land area of the United States (Forman 2000) and 20% of the Netherlands (Reijnen et al. 1995). Areas larger than 100 km² that were unfragmented by roads decreased from 22% to 14% of the total land coverage of the old West German states between 1977 and 1998 (Bundesamt für Naturschutz 1999), and this trend is very likely to continue in most parts of the world (e.g., NRTF 1997).

Although there is now a growing body of evidence of the negative impacts of roads on wildlife (Trombulak and Frissell 2000, Underhill and Angold 2000, Forman et al. 2002, Sherwood et al. 2002, Spellerberg 2002), ecological research has had comparatively little effect on decision making in transportation planning (OECD 2002, UBA 2003). In part, this reflects the fact that, in the face of compelling economic and social arguments for road siting, design, and construction, the effects on ecological values are usually considered of secondary importance (Caid et al. 2002, Bratzel 2005). However, this lack of resonance also relates to the nature of road research itself. Maximizing the impact of road research in the decision-making process requires that (1) the questions addressed by road ecologists be directly relevant to the practical issues of road planning and construction, and (2) road studies be designed so as to have high evidentiary weight. Much of the road research undertaken thus far fails to satisfy at least one, and often both, of these conditions (see the subsection on the current state of road ecology research), with the result that ecological arguments often must appeal to general decision-making principles such as the precautionary principle (Myers 1993, Underwood 1997). These are frequently viewed as unscientific apologetics, especially in the face of compelling economic counterarguments (Foster et al. 2000, Sunstein 2003, Goldstein and Carruth 2004).

In this paper we suggest ways to make road research more relevant and effective by addressing questions of direct management concern and designing studies that have high inferential strength. We begin by identifying the questions in road ecology of most direct relevance to the decision-making process. We then describe how the inferential strength of studies is influenced by study design and extrapolation. We proceed to a methodological standard for road ecology research by specifying, for each of the research questions identified, a hierarchy of experimental designs. We conclude with a discussion of the implications for road ecology researchers, planners, and funding organizations.

2.2 RELEVANT QUESTIONS IN ROAD ECOLOGY

The design and operation of road networks that minimize ecological impacts requires an understanding of how roads affect wildlife populations and how negative effects can be mitigated. These practical needs give rise to five empirical research questions.

2.2.1 QUESTION 1

UNDER WHAT CIRCUMSTANCES DO ROADS AFFECT POPULATION PERSISTENCE?

Despite a fair amount of literature on the effects of roads on animals (reviewed in Glitzner et al. 1999, Jackson 2000, Trombulak and Frissell 2000, Underhill and Angold 2000, Forman 2002), very few studies evaluate the effects of roads at the population level. Most studies either document road mortality or evaluate the effects of roads and traffic on animal movement, neither of which allows strong inference about the impacts on population viability; for example, it is possible that increased reproduction rates counterbalance losses caused by traffic mortality (see the subsection on the current state of road ecology research). Because the extent to which a road affects population persistence may depend on the particular circumstances, it seems likely that answering this question will depend on synthesizing the results from a set of studies conducted under a variety of circumstances.

2.2.2 QUESTION 2

WHAT IS THE RELATIVE IMPORTANCE OF ROAD EFFECTS VERSUS OTHER IMPACTS ON POPULATION PERSISTENCE?

Roads are only one of a suite of anthropogenic stressors to which wildlife populations are exposed. Developing efficient and effective strategies for mitigating population declines requires knowledge of the relative importance of different stressors and the extent to which they interact. This represents an enormous logistical challenge, especially because estimating interactions requires factorial-design experiments.

2.2.3 QUESTION 3

UNDER WHAT CIRCUMSTANCES CAN ROAD EFFECTS BE MITIGATED?

Assuming that roads have negative effects on wildlife populations and that roads contribute substantially to the decline of wildlife populations relative to other impacts, the obvious planning question is the extent to which road effects can be mitigated, and at what economic cost. Mitigation may include modification of road siting, design, and construction as well as the installation of barriers, speed limits, noise screens, and under- or over-road wildlife passageways (Iuell et al. 2003).

Mitigation may not guarantee a viable population, because the starting point of no roads may have already been a nonviable population. Moreover, mitigation may only be partial, but, if it substantially improves population viability, it may be considered successful. The extent to which a particular road effect can be mitigated depends on the particular circumstances, such as the biology of the target species, road characteristics, or neighboring habitat, and the choice and design of the particular mitigation measure (Clevenger 2002). Therefore, as with the first question, it seems likely that answering this question will depend on synthesizing the results from a set of studies conducted under a variety of circumstances.

2.2.4 Question 4

What is the relative importance of the different mechanisms by which roads affect population persistence?

There are four major categories of primary road effects (Van der Zande et al. 1980, Forman 1995, Iuell et al. 2003): (1) mortality from collisions with vehicles; (2) hindrance to movement causing reduced access to resources and mates; (3) disturbance caused by noise, dust, light, and heavy metal pollution, leading to the degradation of habitat quality; and (4) habitat loss caused by disturbance effects in the wider environment and from the physical occupation of land by the road. In addition, road construction is often followed by various indirect effects such as increased human access causing disturbance of breeding sites, increased exploitation via activities such as hunting (McLellan and Shackleton 1988, Kilgo et al. 1998), and the spread of invasive species (Parendes and Jones 2000).

Knowledge of the relative importance of the different road impacts makes it possible to focus mitigation efforts on alleviating the most harmful effects (Osenberg and Schmitt 1996). For example, fencing may effectively mitigate road mortality, but if the major impact is reduced habitat connectivity, fencing may do more harm than good (Jaeger and Fahrig 2004).

Well-designed studies that identify effective mitigation measures can also demonstrate which negative effects of roads are the largest. On the other hand, there are mitigation strategies that are capable of mitigating multiple effects. For example, if a wildlife overpass is shown to be effective, it remains unclear whether this is because of decreased road mortality, increased movement, or both.

2.2.5 Question 5

Under what circumstances do road networks affect population persistence at the landscape scale?

Newly constructed roads add to the existing road network and may affect wildlife at both local and landscape scales (Forman 1995). The scale of an investigation is, among other things, dependent on the area requirements of the species observed, and landscape-scale effects may be particularly relevant for species with large home ranges. Some landscape-scale road effects can be studied at a local scale and extrapolated to the landscape, albeit with attendant extrapolation issues. Other landscape-scale effects must be directly addressed at a landscape scale, for example, questions about how the configuration of road networks affects population persistence. Landscape-scale studies may provide, for example, information about where in the landscape mitigation wildlife overpasses or fences should be placed.

2.3 Sound road ecology

2.3.1 About inferential strength

For any scientific question under study, there is the truth that the experimenter is attempting to uncover and the actual result derived from the experiment. Hence, the key question in any scientific study is: Given a set of results, what is the strength, i.e., validity, of the inference that the hypothesis tested is true or false? The probability for the inference that the result is indeed the truth is associated with a specific level of uncertainty. Low strength of inference means high uncertainty. To counterbalance economic arguments, road ecology studies need to be designed with the highest inferential strength possible, and doing sound road ecology requires study designs with high inferential strength wherever such studies are feasible.

The inferential strength of a study depends on (1) the number of competing hypotheses tested, (2) the study design, and (3) the extent to which one must extrapolate from the context in which the study was conducted to the context of concern, i.e., the particular decision context. Inferential strength increases with the number of competing hypotheses tested because there are always many possible hypotheses consistent with any given experimental result (Chamberlin 1965). With respect to study design, experimental manipulations generally have higher inferential strength than do correlation studies, because of the generally greater ability to control confounding factors. Inferential strength declines with increasing extrapolation, because the greater the extrapolation, the less likely it is that the causal structure of the experimental domain is mimicked in the domain of real interest.

2.3.2 Hierarchy of study designs

We distinguish between two major dimensions of study design: (1) study class and (2) study type. With regard to *study class*, we distinguish between manipulative and nonmanipulative studies. The main difference is that the manipulative study is prospective, looking forward in time, whereas the nonmanipulative study is retrospective, looking backwards in time (Sahai and Sahai 1996). For example, a manipulative study might monitor a population in a location in which a road is planned and then continue to monitor that population during and after road construction. In a nonmanipulative study, data from a site before and after a road was built may be available, and the researcher assesses retrospectively what happened to the population. The main problem with a nonmanipulative study is that some of the information needed, e.g., historical habitat data, may be unavailable or nonexistent. Manipulative studies have higher inferential strength, because the researchers conduct the study knowing that the manipulation, e.g., road construction, will occur. This allows collection of all the relevant data before, during, and after the manipulation. Furthermore, if the researchers are integrated into the road planning project, they may have some control over the design of the manipulation itself.

The *study type* with the highest inferential strength for assessing human impacts on the environment is the before-after-control-impact (BACI) design (Green 1979, Underwood and Chapman 2003). In this design, the sites affected by the human impact (I) are compared with unaffected control sites (C) both before (B) and after (A) some intervention. If the difference in the environmental variable of interest, e.g., wildlife population size, between the control and impact sites is greater after the impact than before, this is strong evidence that the intervention has caused the observed change. BACIs should have replication in space, i.e., several control and impact sites should be studied (Underwood 1992), and replication in time, i.e., the environmental variable(s) should be measured multiple times both before and after the impact (Bernstein and Zalinski 1983, Stewart-Oaten et al. 1986).

In some situations, a complete BACI study is not possible, and there are two partial designs of lower inferential strength. In before-after (BA) designs, a site or sites is studied before (B) and after (A) an impact, but there are no control sites. If the environmental variable, e.g., population size, changes after the impact, that is probably because of the impact. However, because there are no control sites, the possibility that the observed change was caused by something other than the observed impact cannot be excluded (Osenberg and Schmitt 1996).

In the control-impact (CI) design, data exist only for the period after the impact. Affected sites, e.g., sites with roads, are compared with unaffected sites. If control (C) and impact (I) sites differ with regard to the environmental variable, the inference is that this difference is because of the intervention. Clearly, this inference is valid only if control and impact sites are identical in the absence of the intervention, an assumption that cannot be tested, because the BA component is missing (Osenberg and Schmitt 1996). Replication in space of both C and I sites can reduce this problem, but not eliminate it. A correlation study measuring the environmental variable in sites along a gradient of the impact, e.g., road density, represents a common type of CI design.

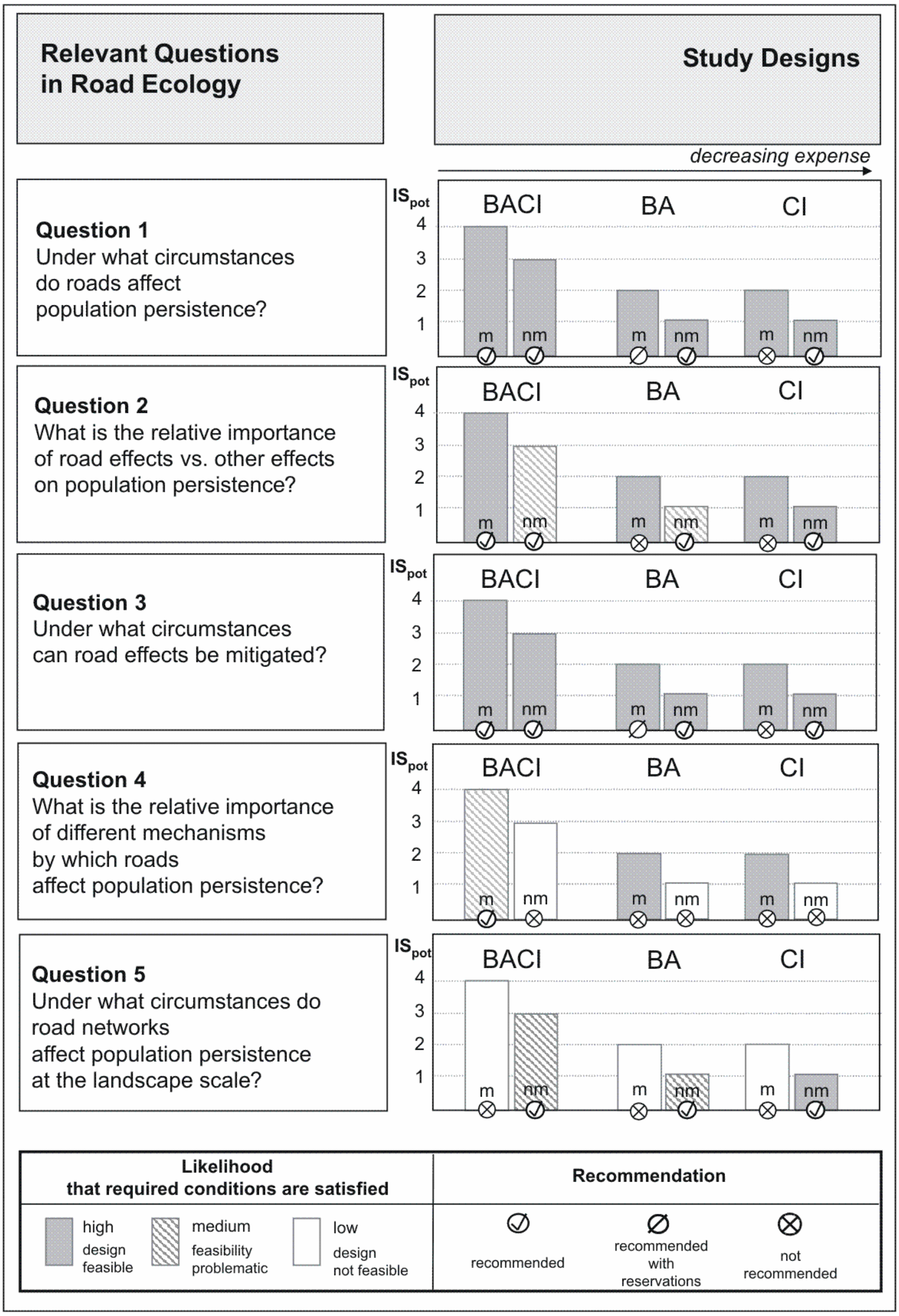

Fig. 2.1*: The Rauischholzhausen-Agenda for Road Ecology: Five key questions and potential study designs. There are two study classes: manipulative (m) and non-manipulative (nm); and three study types: before-after-control-impact (BACI), before-after (BA) and control-impact (CI). Column height indicates inferential strength in ordinal ranks: (1) low, (2) medium, (3) high, and (4) very high. For each design, we provide a qualitative estimate of feasibility (cost, likelihood of required data being available, etc.) and a recommendation based on feasibility and inferential strength.*

Although a replicated, manipulative BACI design is desirable, it cannot be used in situations in which there are no control sites, randomization of sites to treatments is not possible, no data from the period before impacts are available, and/or financial resources are limited. As such, we can develop a rough hierarchy of experimental designs ranked according to their *a priori* inferential strength. Unfortunately, feasibility declines with inferential strength, because the greater the inferential strength of a study, the greater the number of design requirements that must be fulfilled and the number of resources required to fulfill them. Hence, for each design one can also estimate a degree of feasibility depending on costs and the availability of the required data, and a recommendation of expedience based on feasibility and inferential strength (**Fig. 2.1**).

2.3.3 The problem of extrapolation

Rarely do the conditions under which the research was conducted completely match the conditions in which the research results are to be applied, and the greater the extrapolation, the lower the inferential strength of the study. Four main types of extrapolation occur in road ecology:

1. *Spatial extrapolation* occurs when either the research results at one site are used to make inferences about road effects at other sites or the results of a road ecology experiment in a small area, e.g., a 100-ha forest fragment, are used to make inferences about road effects in a larger area measuring, e.g., 1000 km². In the first situation, it may not be possible to conduct the experiment at the sites of interest because of political or logistical concerns. In the second, it might be impossible to conduct the experiment on the required spatial scale because of time and resource constraints.

2. *Temporal extrapolation* occurs when the results of a short-term experiment lasting, e.g., 12 months are used to make inferences about the long-term response, e.g., population persistence. Of particular concern here is the possibility of time lags in the relationship between the impact, i.e., road, and response, i.e., population size (Findlay and Bourdages 2000). In such instances, extrapolation of a weak short-term effect would lead to an underestimate of the effect over the long term. However, temporal extrapolation is often necessary because it is rarely feasible to design a BACI or BA study that will last several decades.

3. *Taxonomic extrapolation* occurs when the results of studies of a single or a few representative focal species, i.e., umbrella species, are used to infer effects on other species or groups of species (Lambeck 1997, Lindenmayer et al. 2002, Caro et al. 2005, Ozaki et al. 2006). This is necessary when either the particular species of interest cannot be studied because of logistical or political constraints, or the objective is to develop principles about the effects of roads on wildlife in general but it is not feasible to study a large number of species.

4. *Endpoint extrapolation* occurs when the ´assessment endpoint´ is far away from the ´measurement endpoint´. The ´assessment endpoint´ is the environmental value of actual interest; here, it is the probability of population persistence. The ´measurement endpoint´ is the response that is actually measured (Suter 1990, 1993). Because population viability cannot be directly measured, the attributes of the population that are known to be related to population viability, such as changes in population size over time, age structure, or number of road-killed individuals, are estimated. Some measurement endpoints are more closely related to the assessment endpoint than others, thereby reducing extrapolation. For example, if the measurement endpoint is population trends over time and the researcher observes a declining trend following road construction, he is more confident in making a prediction about the effect of the road on population persistence than if the measurement endpoint is the number of road-killed animals (see the subsection on the current state of road ecology research).

2.4 EXPERIMENTAL DESIGNS

In this section, we describe a set of studies that vary in their inferential strength and feasibility for each of the focal questions asked above. We begin with question-specific discussions of potential before-after-control-impact (BACI) designs. We then move to a more general discussion of before-after (BA) and control-impact (CI) designs, because the issues surrounding these designs are generic to all the questions.

2.4.1 BEFORE-AFTER-CONTROL-IMPACT DESIGNS

QUESTION 1: UNDER WHAT CIRCUMSTANCES DO ROADS AFFECT POPULATION PERSISTENCE?

There are three general types of road BACIs (**Fig. 2.2**). In the *road-construction* BACI, the population is surveyed before and after road construction at sites at which a road was built and at control sites at which no road was built. In the *road-modification* BACI, the population is surveyed before and after an existing road is modified in some manner, e.g., expansion from two to four lanes or the installation of street lights, at road sites at which the modification occurred and at control road sites at which no modification occurred. In the *road-removal* BACI, the population is surveyed before and after road removal or closure at sites at which removal occurred and at control sites at which the road was not removed.

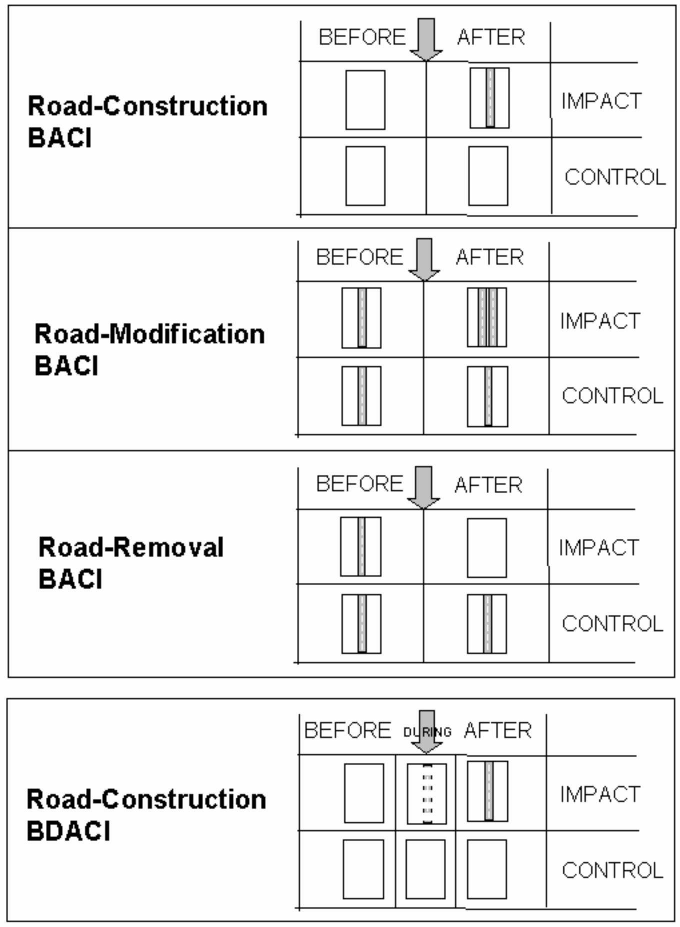

***Fig. 2.2**: Three types of Road-BACIs (before-after-control-impact study designs). An elaboration of the Road-Construction-BACI is the BDACI design (before-during-after-control-impact), where effects may be evaluated during the road construction phase.*

A subtype of the road-construction BACI is the *road-construction BDACI* (before-during-after-control-impact). Here, the population is surveyed before, during, and after road construction at sites at which a road was built and control sites at which no road was built. This study increases our knowledge about the (ir)reversibility of the effects that might occur during the road-construction phase, but may disappear after construction, e.g., the effects of noise from road construction.

All types of road BACIs require at least two sets of replicated sites (**Fig. 2.3**): (1) impact sites with road construction, modification, or removal and (2) control sites without such interventions. Control sites should be located outside the road effect zone of the planned road construction, modification, or removal (Reijnen and Foppen 1994) and should be as similar as possible to the impact sites with regard to land use, species composition and abundance, and particularly the habitat requirements of the species of interest. A measurement endpoint closely related to population persistence, e.g., abundance of the species of interest, should be sampled at multiple times before and after the intervention in both the control and impact sites. The population should be surveyed for long enough following the intervention to capture possible lag effects. This design can be implemented as a manipulative or a nonmanipulative BACI (**Fig. 2.1**).

Question 2: What is the relative importance of road effects on population persistence?

A modified design for a road-construction BACI is desirable for determining the relative importance of roads vs. some other impact. For example, if the other impact is pesticide use, the study should include (1) control areas without either impact, (2) impact sites with road construction but without pesticide use or with reduced use, and (3) impact sites without road construction but with pesticide use (**Fig. 2.4**). Additional sites with both roads and pesticide use address the issue of potential interactions between stressors. If, for example, roads have a larger effect on wildlife populations than does pesticide exposure, the difference between before and after should be greater for the contrast between control and road-impact sites than for the contrast between control and pesticide-impact sites. Impacts should ideally commence simultaneously, to avoid the confounding effects of different characteristic response times for different stressors. Control and impact sites should be replicated, with multiple sampling times both before and after the intervention(s).

This design can be implemented as a manipulative or a nonmanipulative BACI (**Fig. 2.1**). However, the nonmanipulative study requires sites at which the road impact and the pesticide impact commenced simultaneously, and for which data were available on the population both before and after the impacts. Because it would be extremely difficult to find such sites, this design has low feasibility (**Fig. 2.1**).

Question 3: Under what circumstances can road effects be mitigated?

We recognize three general types of mitigation BACI designs (see also **Fig. 2.2**). In the *mitigation-construction* BACI, the population is surveyed before and after the construction of a mitigation measure at sites at which a mitigation measure was built and at sites without mitigation measures. In the *mitigation-modification* BACI, the population is surveyed before and after an existing mitigation strategy is modified in some manner, at both the control and impact sites. In the *mitigation-removal* BACI, the population is surveyed before and after the removal of a mitigation measure, at both control and impact sites.

There are three possible mitigation-construction BACI designs. The first determines whether new roads that are built with mitigation measures in place have a smaller effect on the population of interest than do roads built without mitigation measures. Here, populations are compared at (1) control sites with no roads, (2) control sites with roads with no mitigation measures, and (3) ´impact´ sites with roads and mitigation measures. The measurement endpoint is sampled before and after the roads with and without mitigation measures are constructed. The results will be useful in designing new roads.

The second design determines whether mitigation measures can restore a population to viability after a road has already been affecting it for some time. This can be answered by the mitigation-construction BDACI (before-during-after-control-impact). At the impact site, the road is first constructed without a mitigation measure, and then some time later a mitigation measure is added (**Fig. 2.5**). The population is sampled before road construction, after road construction without mitigation, i.e., during sampling, and after the addition of the mitigation measure. This design requires two control sites: one with no road and one with a road without mitigation. The before-during comparison provides information on the size of the road effect; the during-after comparison provides an estimate of the extent to which the road effect is mitigated. This design is feasible as a manipulative or nonmanipulative study. However, it is particularly susceptible to problems of time-scale extrapolation (see the subsection on the problems of extrapolation), because both road and mitigation effects are likely to show lagged responses.

A simpler mitigation-construction BACI is obtained by choosing only sites that already have roads on them and sampling the population before and after the addition of the mitigation measure. This design will indicate whether the mitigation measure is effective, but not whether the road effect is fully mitigated. However, it can provide valuable information for de-fragmentation programs aimed at mitigating the impacts of the current road network (Van der Grift 2005).

It is also possible to compare the effectiveness of two mitigation measures such as fences and passageways. In addition to the sites mentioned above, sites with the road and the second mitigation measure are also required. Additional sites with the road and both mitigation measures would allow evaluation of the combined effects of both mitigation measures.

QUESTION 4: WHAT IS THE RELATIVE IMPORTANCE OF THE DIFFERENT MECHANISMS BY WHICH ROADS AFFECT POPULATION PERSISTENCE?

Here the general approach is to establish conditions under which only a single mechanism is possible at one time or in one context. To illustrate, we discuss study designs that distinguish the relative effects of mortality vs. movement barriers, but the same designs could be applied to any pair of effects, e.g., mortality vs. disturbance, disturbance vs. movement barrier, one type of disturbance vs. another type of disturbance, etc. Answering this question requires an elaborate and rather artificial manipulative BACI design. It is not possible to answer this question in any type of nonmanipulative study (**Fig. 2.1**).

The BACI design requires five sets of replicated sites (**Fig. 2.6**): (1) control sites containing no roads and no movement barriers at which movement is unhindered and there is no road mortality, (2) sites with no roads that incorporate fences as movement barriers, (3) sites with no roads and no fences at which mortality is simulated by removing individuals from the population at a rate equal to estimated traffic mortality, (4) sites with no roads that incorporate fences at which mortality is again simulated by removing animals, and (5) sites with roads but no fences and no simulated mortality.

Measurement endpoints are assessed before and after treatment at all sites. The magnitude of the road effect mechanism of mortality or barrier is estimated as the difference between the control sites (type 1) and the sites with no roads in which the road effect is simulated (types 2 and 3). Including sites with both simulated effects (type 4) makes it possible to estimate their combined effect. Finally, the difference between type-4 sites and the sites with an actual road present (type 5) allows us to estimate the size of all additional road effects such as traffic noise and habitat disturbances.

This design is very difficult to implement and therefore has low feasibility (**Fig. 2.1**). First, because the different mechanisms of mortality and barrier, in our example, may take different lengths of time to affect population persistence, the populations will need to be monitored for a sufficient time period to estimate their relative effects (Jaeger and Fahrig 2001). Second, simulating road mortality requires a pilot study to estimate road mortality and its demographic effects. Third, because roads are generally not complete barriers to movement, the permeability of the barrier simulation will need to be controlled experimentally by moving some animals across the fence. This in turn will require another pilot study designed to estimate movement rates. Finally, because the mechanisms related to mortality and barrier effect will vary with road type, traffic volume, and season, both pilot studies and the BACI study itself should be conducted under a range of different conditions.

QUESTION 5: UNDER WHAT CIRCUMSTANCES DO ROAD NETWORKS AFFECT POPULATION PERSISTENCE AT THE LANDSCAPE SCALE?

At the landscape scale, three types of nonmanipulative BACIs are possible (see also **Fig. 2**): a *road-construction* BACI in which the density of the road network has increased over time, a *road-modification* BACI in which road density has been constant but traffic volume has increased or network configuration has changed, and a *road-removal* BACI in which roads have been removed from the network.

The road-construction BACI requires at least two sets of sites (**Fig. 2.7**): (1) impact sites at which the road network has increased over time and (2) control sites with no increase in network density over time. The sites and the landscapes should be as similar as possible, particularly with regard to the habitat requirements of the species observed. Control and impact sites should be replicated, and population data must be available before and after the changes to the road network at all the sites, most likely from existing long-term, large-scale programs for monitoring wildlife populations.

Manipulative experiments will generally not be possible on the scale of whole road networks (**Fig. 2.1**). Non-manipulative BACIs will have low feasibility for two main reasons: (1) the difficulty in finding replicated control and impact landscapes in which other factors affecting population persistence, e.g., extension of settlements and agricultural development, are similar over the entire time span of the study, and (2) the requirement that there be reasonably accurate, systematic wildlife population data extending over the entire spatial and temporal span of the study. The larger the required landscapes because of, for example, large dispersal distances of target species, the more difficult it will be to find comparable non-overlapping landscapes and wildlife data.

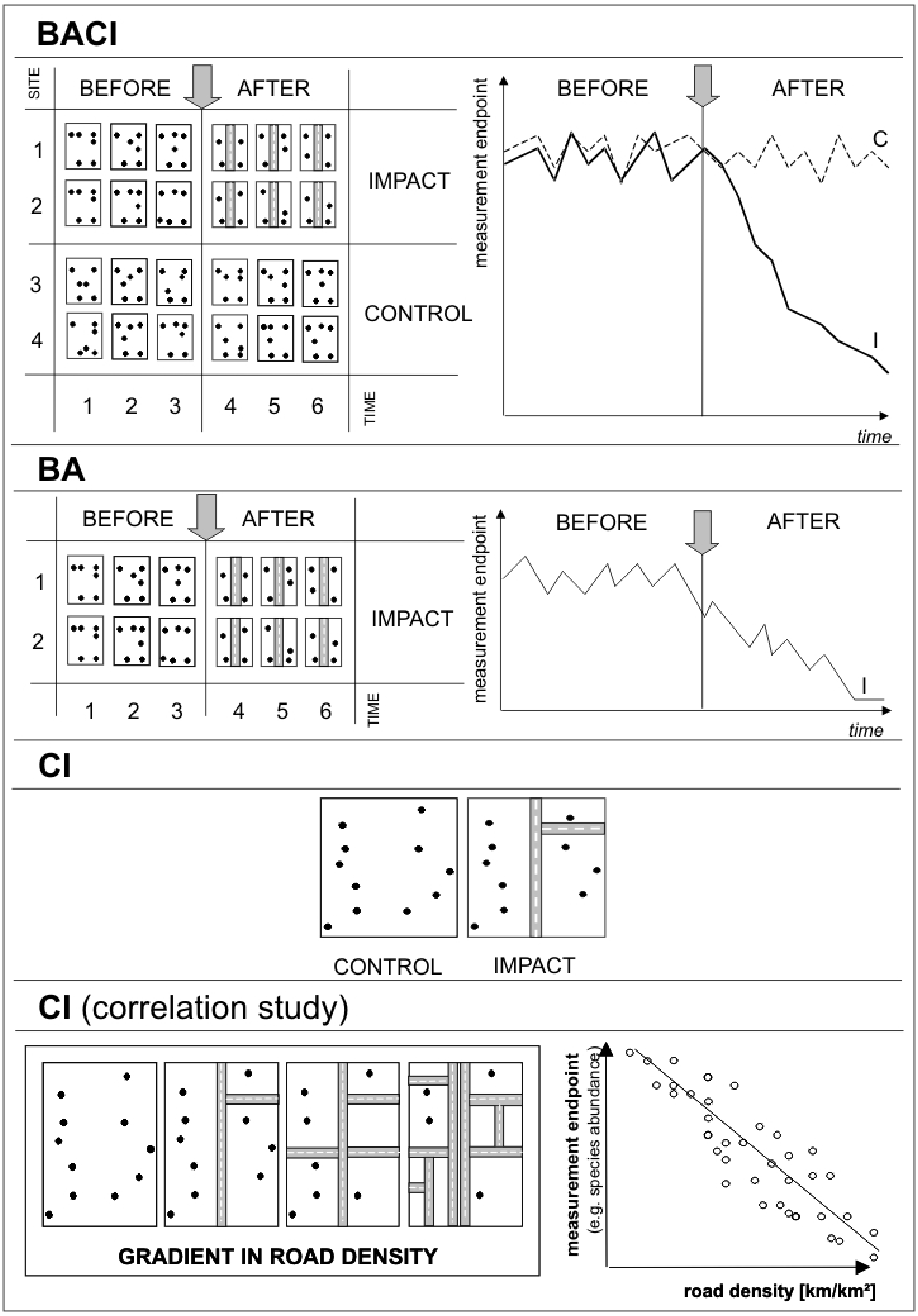

Fig. 2.3: Study designs and hypothetical results for question 1: "Under what circumstances do roads affect population persistence?" We use a Road-Construction-BACI as example (see ***Fig. 2.2****). Study type is before-after-control-impact (BACI), before-after (BA) and control-impact (CI). The correlation study is a subtype of the CI design.*

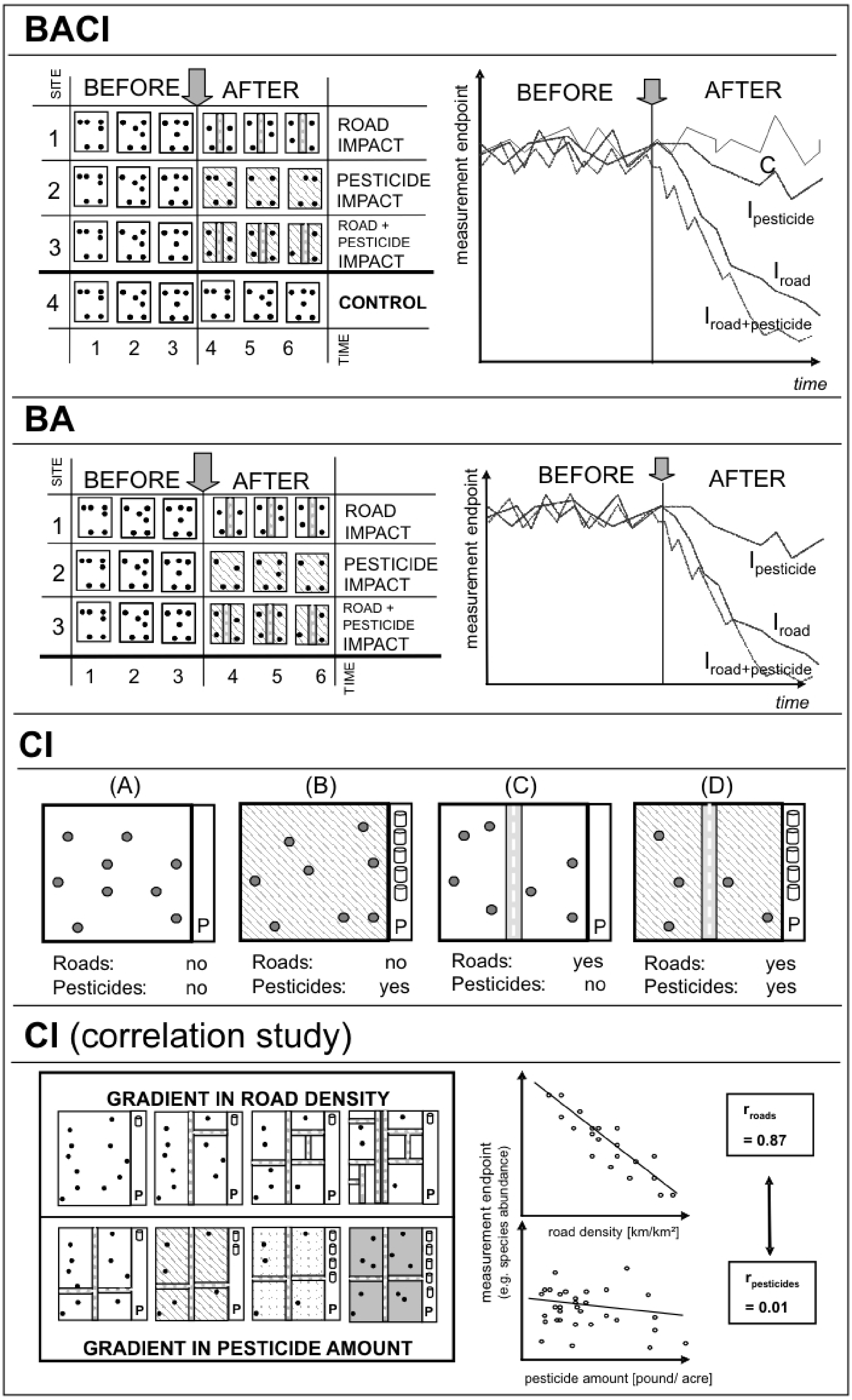

Fig. 2.4*: Study designs and hypothetical results for question 2: "What is the relative importance of road effects on population persistence?" Study type is before-after-control-impact (BACI), before-after (BA) and control-impact (CI). The correlation study is a sub-type of the CI design.*

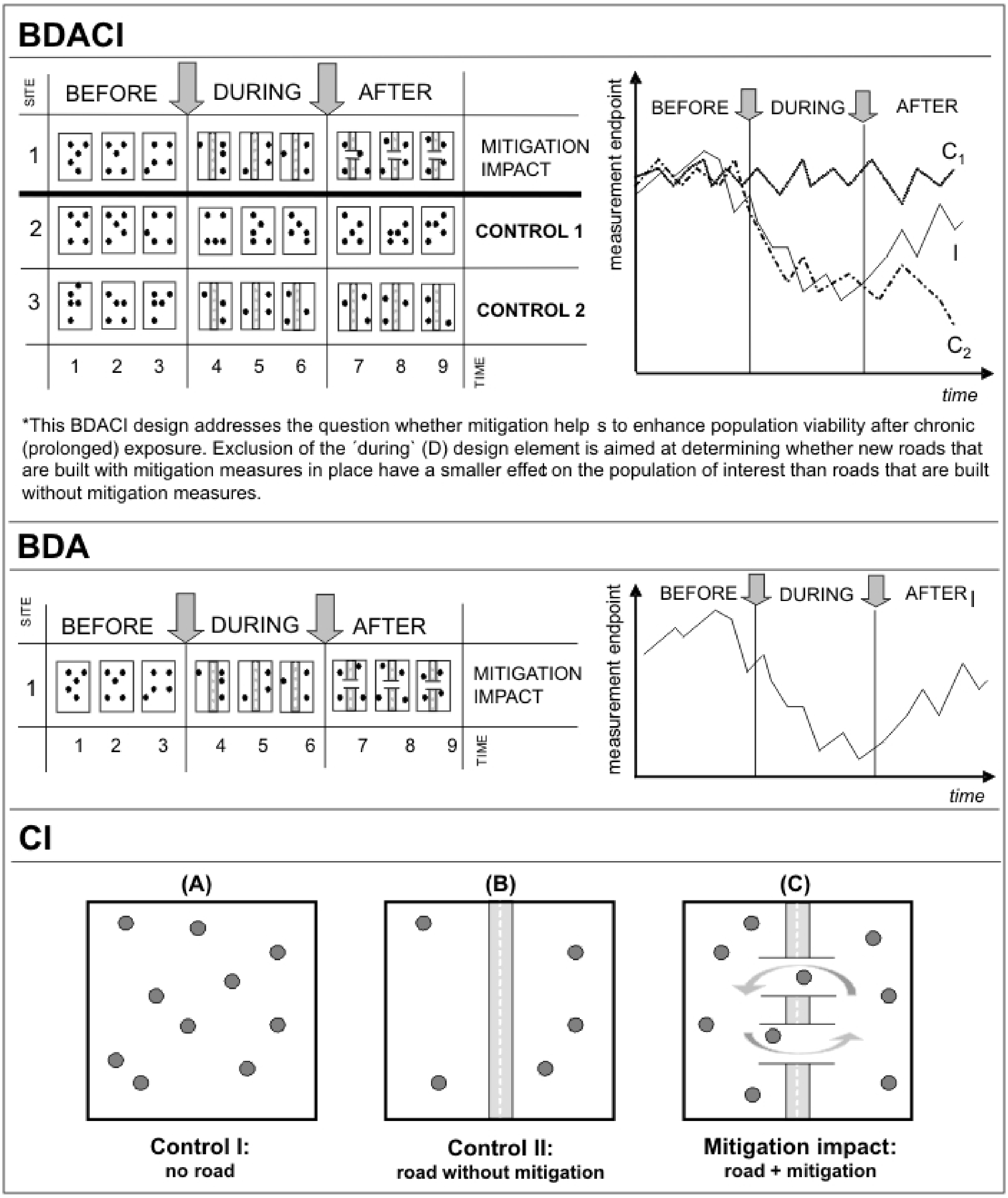

Fig. 2.5: Study designs and hypothetical results for question 3: "Under what circumstances can road effects be mitigated?" Study type is before-during-after-control-impact (BDACI), before-during-after (BDA) and control-impact (CI).

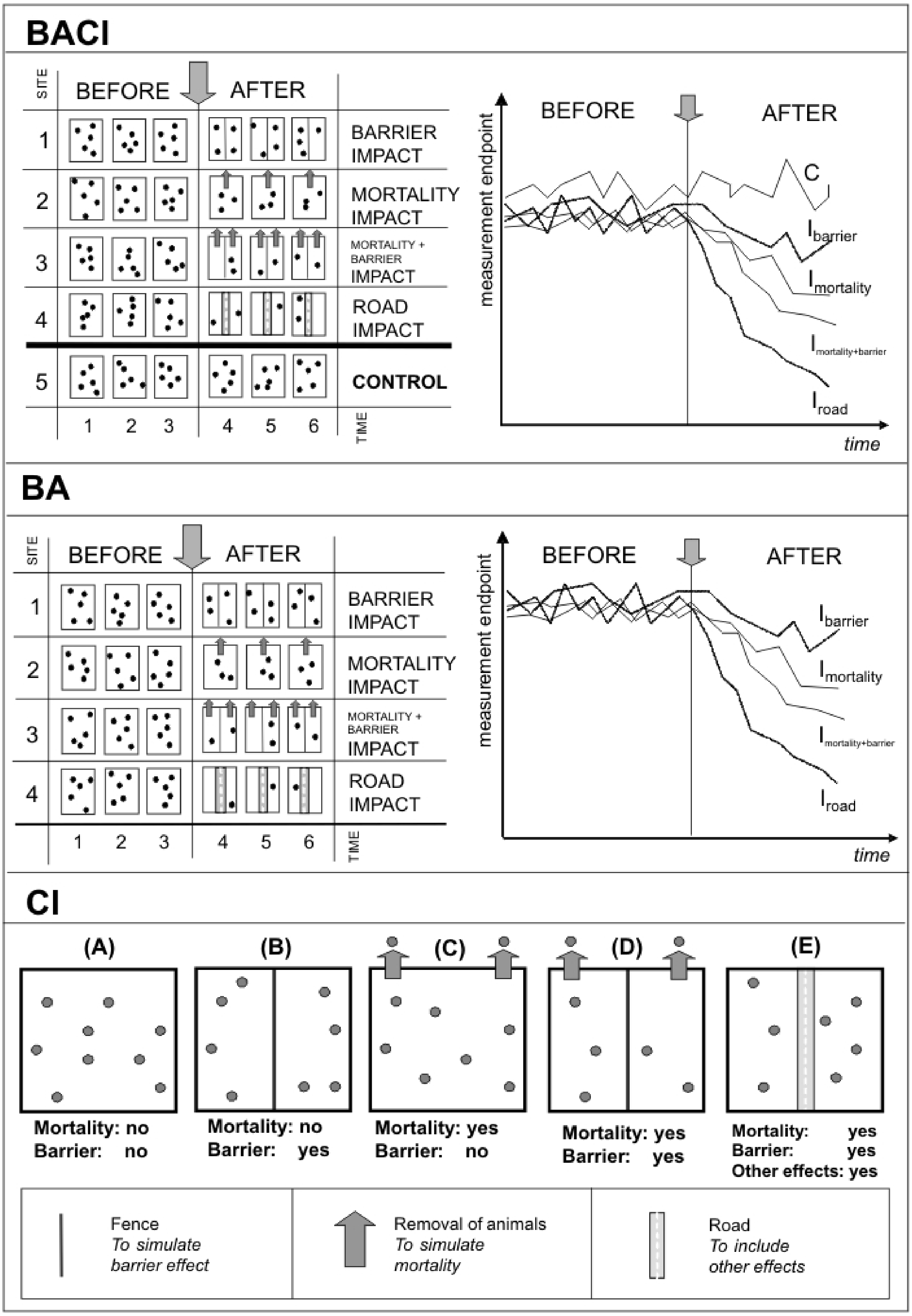

***Fig. 2.6**: Study designs and hypothetical results for question 4: "What is the relative importance of different mechanisms by which roads affect population persistence?" Study type is before-after-control-impact (BACI), before-after (BA) and control-impact (CI).*

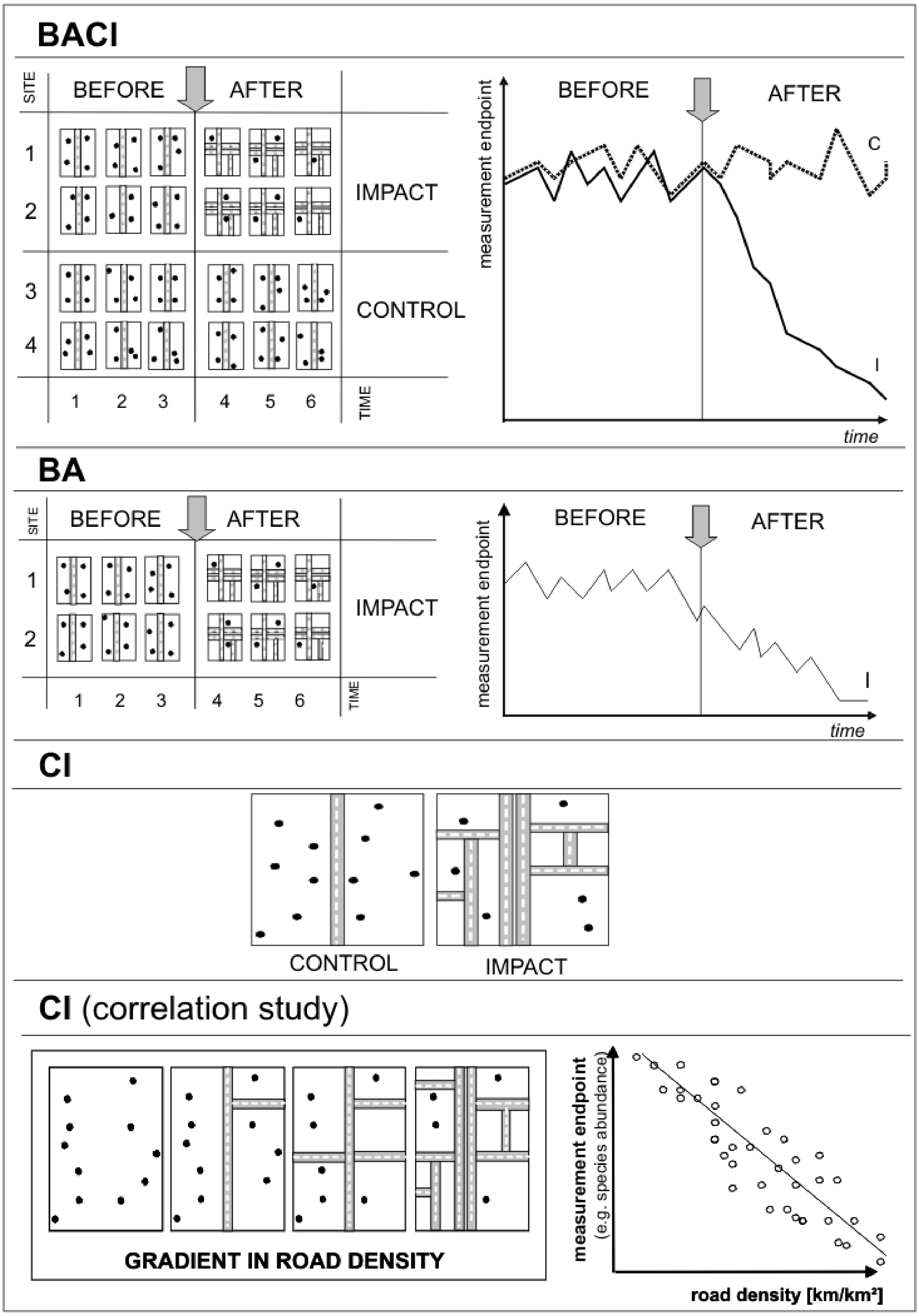

***Fig. 2.7**: Study designs and hypothetical results for question 5: "Under what circumstances do road networks affect population persistence at the landscape-scale?" Study type is before-after-control-impact (BACI), before-after (BA) and control-impact (CI). The correlation study is a subtype of the CI design.*

2.4.2 BEFORE-AFTER DESIGNS

If appropriate control sites for a BACI are lacking, a before-after (BA) design may be used. Here, substantial differences in the selected measurement endpoints before and after the intervention, e.g., road construction or mitigation measures, indicate an effect on population persistence (see the subsection on the hierarchy of designs).

A manipulative BA design gives some control over the design of the manipulation itself, because the researcher is conducting the study in the knowledge that the manipulation, e.g., road construction, will occur. We do not recommend manipulative BA designs, because, given the expense and time required for them, it is worth including control sites and performing the full BACI to realize the maximum possible inferential strength, especially for questions requiring a large amount of prospective planning (Questions 2 and 4). However, there may be situations in which there is no choice of impact sites, e.g., because of political concerns, and appropriate control sites may simply not be available (Questions 1 and 3). In this case a manipulative BA is the design of choice. At a large scale, a manipulative BA is not feasible (Question 5).

Nonmanipulative designs are those for which the researcher has information on endpoints collected both before and after the intervention but had no hand in the intervention itself. Because of the retrospective character of the study, it is possible that no appropriate control sites can be found, and in this case a nonmanipulative BA is the design of choice (**Fig. 2.1**). The feasibility of nonmanipulative designs might be problematic or impossible for the same reasons outlined above in the description of BACI designs.

2.4.3 CONTROL-IMPACT DESIGNS

If preintervention data are unavailable, a control-impact (CI) design can be used in which the population is surveyed in (1) sites with and without a road present (**Fig. 2.3**), (2) sites representing two uncorrelated gradients of road density and another stressor such as pesticide use (**Fig. 2.4**), (3) sites with and without one or more mitigation measures (**Fig. 2.5**), and (4) sites spanning a gradient of road densities or traffic volumes, i.e., a correlation study (**Fig. 2.7**).

We do not recommend manipulative CI studies for any focal question, simply because, if a prospective manipulative CI is possible, there is little excuse for not monitoring both before and after construction, in which case the study incorporates all elements of the BACI design (**Fig. 2.1**). At a large scale, a manipulative CI is simply not feasible. The nonmanipulative CI design is recommended for all of the above questions except the one related to the relative importance of the different mechanisms by which roads affect population persistence, because, retrospectively, it is highly unlikely that the artificial conditions required for the design of that question will be satisfied (**Fig. 2.6**).

2.5 Discussion

2.5.1 Current state of road ecology research

Past studies in road ecology generally have low inferential strength for two main reasons. First, the usual measurement endpoints are typically well removed from the quantity of interest, namely, population viability and persistence. One of the most common measurement endpoints is movement across roads (e.g., Mader 1984, Merriam et al. 1989, Brody and Pelton 1989) and/or through mitigation structures (e.g., Clevenger and Waltho 2000, 2005). Although movement is an important component of population dynamics, its predictive value with respect to population persistence is low. For example, although roads have a negative effect on the movement of small mammals (Mader 1984), the density of the population of small mammals is sometimes positively associated with roads, possibly because of the negative effect of roads on predator populations (T.D.M. Rytwinski and L. Fahrig, unpublished manuscript) or alterations in road site habitats in favor of small mammals. Therefore, inferring impacts on population persistence from the effects on movement rates is fraught with uncertainty. Similar problems of inference arise from estimates of road-induced mortality, another common measurement endpoint in road ecology (e.g., Lodé 2000, Baker et al. 2004). Inferring effects on population persistence from the numbers of road-killed individuals is very tenuous and requires information on population size, variability, and the likelihood of compensatory mechanisms such as reductions in other mortality sources or increases in reproduction in response to road mortality.

Second, most studies are control-impact (CI) designs without before-after (BA) data (e.g., Ballon 1986, Mech et al. 1988, Findlay and Houlahan 1997, Clarke et al. 1998, Vos and Chardon 1998, Findlay and Bourdages 2000, Carr and Fahrig 2001). Although these studies have produced some suggestive results, the inferential strength of CI designs is always lower than that of BACI designs and usually lower than that of BA designs. Without data on endpoints from before the impact, one cannot rule out the hypothesis that the difference between the control and impact sites is because of preexisting differences among the sites that are unrelated to the road.

A science that is built on studies of generally low inferential strength is problematic. The lower the inferential strength of the studies, the more likely that the collection of such studies will produce apparently conflicting results even if the underlying hypotheses are generally true. Not only does this create uncertainty, but it also results in considerable effort being expended to "resolve" apparently contradictory results, when in fact the contradiction may simply reflect incorrect inferences arising from poor experimental designs. This was underlined by Danielson and Hubbard (1998) in their review of three studies that evaluated the effectiveness of Swareflex reflectors in reducing vehicle wildlife collisions; one of these studies used a BACI design (Gladfelter 1984), and the other two used a CI design (Schafer and Penland 1985, Reeve and Anderson 1993). Here, the contradictory results that resulted from poor study designs caused state transportation agencies to expend considerable resources repeating the research or implementing mitigation measures that had not really been proven to be effective (Danielson and Hubbard 1998).

Inferential strength matters, in particular, in environmental impact assessment (EIA) studies, in which research "meets" decision making. Decision makers are legally obligated to commission EIAs to estimate the potential impacts of proposed roads and the extent to which these expected impacts can be mitigated. A review was conducted in the UK to determine if the EIAs for proposed roads met the minimum standard of scientific rigor necessary to make useful inferences (Treweek et al. 1993, Byron et al. 2000). It highlighted certain shortcomings in the EIA process. For instance, the studies did not differentiate between the relative importance of different road effects, and they did not collect data from the period after the construction of roads or mitigation measures. The reviewers concluded that EIA studies were not of an appropriate type to capture relevant ecological information. However, decisions about road siting and mitigation measures are based on the results of these studies, and the lower the inferential strength, the less certain one can be that expected impacts will indeed be observed and can really be mitigated. The reviewers concluded that the scientific basis needed to be improved, because much survey effort is wasted on studies generating information that contributes little to the decision-making process.

2.5.2 Implications

The experimental design issues raised in this paper have implications for research scientists, scientific funding organizations, planners, and decision makers. For researchers, the implications are straightforward: when designing studies on the effects of roads or the mitigation of road effects, they should strive toward the maximum possible inferential strength, given existing constraints. When practical considerations dictate a study design of lower inferential strength than is desirable, it is important that the results from such studies be interpreted with caution, and that the resulting conclusions be appropriately tempered. It is possible that definitive experiments with high inferential strength carried out over the relevant temporal and spatial scales cannot feasibly be undertaken by individual researchers. Instead of continuing to undertake isolated studies in the hope that the sheer volume of them may compensate for the absence of definitive studies, it seems advisable to combine forces, resources, and expertise in a study that has high inferential strength and permits generalizable and robust conclusions.

For scientific funding organizations, the implications are equally straightforward: there is, in general, a positive correlation between the inferential strength of a study and the resources required to carry it out. As such, funding agencies cannot insist that the same study meet the mutually exclusive goals of high inferential strength and low cost. Moreover, we argue that an investment in a good experiment is actually more cost-effective than a series of "shot-in-the-dark" attempts to fix a problem. An efficient experiment is not simply defined as a cheap experiment; rather, it is an experiment that derives the required information for the least expenditure of resources

(Barker 1994). We have identified feasible study designs of reasonably high inferential strength (**Fig. 2.1**); a funding agency will maximize the scientific value and cost-effectiveness of research by giving high priority to these types of studies. The demand for such studies is of some urgency, because, even with a willingness to commit resources, future research may be limited by the fact that few landscapes exist in which to undertake the necessary experiments.

For planners and decision makers, the most important issue is that constraints on feasibility and costs will necessarily limit inferential strength. For example, for questions concerned with landscape-scale ecological effects and long-term consequences, the inferential strength of any feasible study will always be comparatively low. Study designs of lower inferential strength, such as the nonmanipulative CI design, may be the best one can do in these situations. Consequently, it is inevitable that the uncertainty associated with any conclusion will necessarily be high. Nevertheless, the most pressing policy and management issues are generally not at the local, but at the landscape scale (National Research Council 2005). It is a cruel irony in road ecology that, the more important the question, the more uncertainty is associated with the answers that road science will be able to provide.

For road ecology, and especially those issues relevant to landscape-level planning and management, a strong weight of evidence, i.e., scientific proof, is unattainable in practice, and to insist upon it is tantamount to discounting all the scientific research that is likely to be conducted now or in the foreseeable future. Nevertheless, decisions must be made. For such questions, the standard of proof required for consideration in the planning process must be comparatively low, and decision makers must embrace general normative decision-making principles and approaches for judgment under uncertainty. Examples are the precautionary principle and the establishment of quantitative limits or objectives to limit road density or the degree of landscape fragmentation; both require transdisciplinary discussions among scientists, the public, and decision makers (see Jaeger and Scheringer 1998, Böschen et al. 2001, Jaeger 2002). The task of the road ecologist is to provide scientific answers with the highest inferential strength possible; the task of decision makers is to recognize and make decisions in the face of the inherent limitations and uncertainties in these answers.

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Chapter 3. Landschaftszerschneidung in Hessen

Entwicklung, Vergleich zu Baden-Württemberg und Trendanalyse als Grundlage für ein landesweites Monitoring

Roedenbeck IA, Esswein H, und Köhler W. 2005. Landschaftszerschneidung in Hessen. Entwicklung, Vergleich zu Baden-Württemberg und Trendanalyse als Grundlage für ein landesweites Monitoring (mit Kartenbeilage). *Naturschutz und Landschaftsplanung* 37(10): 293-300.

Zusammenfassung

Die Siedlungs- und Verkehrsfläche hat in Deutschland in den vergangenen Jahren weiter zugenommen, weil gesellschaftspolitisch der Bedarf nach einer hohen Mobilität und globaler Vernetzung besteht. Die ökologischen Negativfolgen für Arten und Lebensräume sind jedoch vielfältig und seit langem bekannt. Aus ökologischer Perspektive müssen dringend Handlungsziele definiert werden, um die Flächenzerschneidung in Zukunft zu begrenzen. Jaeger (2000) entwickelte den Zerschneidungsindex Effektive Maschenweite und legte hiermit einen handhabbaren Umweltindikator vor, der derzeit in mehreren Bundesländern in Gebrauch ist, um den Zustand und die Entwicklung der Flächenzerschneidung zu dokumentieren.

Die vorliegende Untersuchung liefert in einer landesweiten, quantitativen Analyse konkrete Zahlen zur Landschaftszerschneidung in Hessen und zieht einen ersten Vergleich zu der Entwicklung in Baden-Württemberg. Beschrieben werden die historische Siedlungsexpansion und die Verdichtung des hessischen Verkehrswegenetzes in sechs Zeitschritten von 1930 bis 2002. Die Ergebnisse zeigen, dass die Zerschneidung seit 1930 kontinuierlich zugenommen hat. Eine differenzierte Betrachtung der 26 Landkreise und 59 Naturräume weist starke räumliche Unterschiede aus. Der heutige Zustand verdeutlicht die Besiedlungsgeschichte entlang der Flusstäler und -auen, wohingegen die hessischen Mittelgebirgslagen später besiedelt wurden und bis heute am wenigsten zersiedelt sind. Mit den vorliegenden Zahlen sind weiterführende Untersuchungen zu den langfristigen Effekten von Flächenzerschneidung auf das Vorkommen von Tierarten möglich. Sie sind eine wichtige Datengrundlage für ein landesweites Monitoring der Zerschneidung und ermöglichen eine Erfolgskontrolle von politischen Maßnahmen, um die Landschaftszerschneidung in Zukunft einzuschränken.

Abstract

Landscape fragmentation in the Federal State of Hesse - Development, comparison to Baden-Württemberg, and trend analysis as a base for a state-wide monitoring. *The latest past has seen a further increase of the total area of settlements and road networks in Germany due to increasing social needs for high mobility and global networks. The negative ecological effects on species and habitats, however, are diverse and well-known. From an ecological perspective targets for action need to be defined to limit further fragmentation. Jaeger (2000) developed an index of fragmentation, effective mesh size, hence defining a manageable environmental indicator. Meanwhile this index has been applied in several states to document the present situation and the historical development of landscape fragmentation.*

Here we provide concrete quantitative data about landscape fragmentation in Hesse, and we draw a first comparison to the situation in Baden-Württemberg. We describe the historic expansion of settlements and densification of the road network in Hesse in six stages from 1930 to 2002. Results show that fragmentation has continuously increased since 1930. A detailed survey of the 26 administrative districts and 59 natural landscapes reveals significant spatial differences. The present situation reflects settlement history along river valleys and flood plains, whilst low mountain ranges were colonised later and still show the lowest degree of fragmentation today. Based on our findings continuative analysis are possible on the long-term effects of landscape fragmentation on wildlife populations. The data in hand is an important base for a state-wide monitoring of fragmentation enabling evaluation of political measures intended to restrict further landscape fragmentation in the future.

3.1 EINLEITUNG

Komfortabel und großflächig lautet bisher das Motto der Verkehrsplaner. Die Siedlungs- und Verkehrsfläche in Deutschland hat im Jahr 2003 um 0,8% zugenommen (UBA 2004) und täglich wurden im selben Jahr rund 93ha Fläche neu bebaut (Statistisches Bundesamt 2004). Die Gründe für den Anstieg sind vielfältig. Zum einen steigen die Ansprüche an Infrastruktur wie privates Wohneigentum und Gewerbefläche. Die damit einhergehende, fortschreitende Zersiedlung der Landschaft wird vor allem am Rand der Großstädte spürbar. Zum anderen steigt der Bedarf nach einer komfortablen Mobilität. Eine hohe räumliche Vernetzung von Ballungsgebieten wird angestrebt und Naturgebiete sollen für Naherholungszwecke gut erreichbar sein. Diese hoch bewerteten Ziele einer nach Globalisierung strebenden Gesellschaft fordern eine Erweiterung und den Ausbau des Verkehrsnetzes.

Dem gegenüber steht die Erkenntnis, dass aus der zunehmenden Landschaftszerschneidung ökologische Negativfolgen entstehen. Durch die Verdichtung des Verkehrswegenetzes und die Ausdehnung der Siedlungsfläche sind Landschaftsräume bedroht, die wildlebenden Tier- und Pflanzenarten als Lebensräume dienen. Sie werden zerkleinert, voneinander isoliert und können zuletzt durch vollständige Überbauung verloren gehen. Neben dem Habitatverlust und der Isolation von Tierpopulationen sind Verkehrstrassen von diversen anderen Störfaktoren begleitet. Nach Jaeger (2001) betreffen die Folgen der Landschaftszerschneidung folgende Bereiche: Bodengefüge und -bedeckung, Kleinklima, Immissionen, Wasserhaushalt, Flora und Fauna, Landschaftsbild und Erholungsqualität.

Gesellschaftliche Ansprüche stehen hier also ökologischen Erfordernissen diametral entgegen. Eine nachhaltige Landnutzung – Leitbegriff der aktuellen Umweltdiskussion – würde jedoch eine gleichberechtigte Berücksichtigung ökonomischer, sozialer und ökologischer Zielvorstellungen bedeuten. Da dies nicht gegeben ist, erklärte der deutsche Bundestag in seiner Nachhaltigkeitsstrategie die Flächenzerschneidung zu einem „strukturellen gesellschaftlichen Problem“ (Deutscher Bundestag 1998).

Doch in der öffentlichen Diskussion beginnt sich langsam die Erkenntnis durchzusetzen, dass der Ausbau weiterer Infrastruktur auch zu einem finanziellen und sozialen Risiko für Bund, Städte und Gemeinden anwachsen kann. Permanent werden neue Gewerbe- und Siedlungsgebiete erschlossen, deren Bedarf vielfach nicht mehr gegeben ist. Innenstädte veröden, die Fixkosten für die Infrastruktur bleiben aber gleich und müssen von immer weniger Stadtbewohnern gezahlt werden. Im Ergebnis dieser Entwicklung wird befürchtet, dass in den Innenstädten zunehmend Problemquartiere entstehen, weil eher sozial Schwache zurückbleiben (UBA 2004). Die Forderung des Naturschutzes, Freiräume als Lebensräume für Wildtiere zu erhalten, steht somit nicht mehr alleine. Auch aus sozialer Sicht besteht heute der Bedarf, das Wachstum der Siedlungs- und Verkehrsflächen einzudämmen.

Es ist dringend notwendig, dass Entscheidungsträger in Politik und Verwaltung Handlungsziele für die Einschränkung der Flächenzerschneidung definieren. Zur

Gewährleistung der Zieleinhaltung müssen diese Handlungsziele auch überprüft werden können. Weil Entscheidungsträger in ihrem Bemühen, möglichst effiziente Politiken zu formulieren, auf verlässliche Informationen angewiesen sind, müssen Maßzahlen und Indikatoren zur Verfügung stehen, mit denen Landschaftszerschneidung messbar ist.

Das Bundesamt für Naturschutz untersucht in regelmäßigen Abständen den Flächenverbrauch, indem unzerschnittene Räume über 100 qkm ausgezählt werden. Die Entwicklung in Flächen, die kleiner als 100 qkm sind, wird durch diese Methode allerdings nicht erfasst. Es existieren verschiedene andere Studien, die Maßzahlen zur Flächenzerschneidung einführen (Grau 1998). Doch keine dieser Maßzahlen konnte sich im wissenschaftlichen Diskurs oder in der praktischen Anwendung durchsetzen.

Erst Jaeger (2002) gelang es, eine Maßzahl zu etablieren, die alle Anforderungen an einen handhabbaren Zerschneidungsindex mit sich bringt. Die von ihm entwickelte *Effektive Maschenweite* wurde erstmals im Gebiet „Raum Kreuzung Schweizer Mittelland" (Müller et al. 1998) und im Strohgäu in Baden-Württemberg (Jaeger 1999) angewendet, um den Zustand der Landschaftszerschneidung zu bewerten. In Baden-Württemberg wurden Zustand und Entwicklung der Landschaftszerschneidung erstmals landesweit protokolliert (Esswein et al. 2002). Die Vorteile der Methode waren offensichtlich und die Bundesländer Sachsen (LfUG 2002), Bayern (Esswein et al. 2004b), Hessen (Esswein et al. 2004a u. b), Schleswig-Holstein (Neumann-Finke 2004) und Thüringen (Voerkel 2005) folgten dem Beispiel.

Es kann nur positiv bewertet werden, dass immer mehr Bundesländer die Notwendigkeit erkennen, den Zustand der landeseigenen Flächenzerschneidung zu dokumentieren. Nur mit einer einheitlichen Datengrundlage, die durch Anwendung der gleichen Methodik geschaffen wird, kann in Zukunft ein flächendeckendes, einheitliches Monitoring der Landschaftszerschneidung möglich sein.

Bisher sind die Gemeinden zwar verpflichtet, im Rahmen der Bauleitplanung mit Grund und Boden sparsam umzugehen, können aber im Rahmen der Abwägung anderen Belangen Vorrang geben. Und so kann auch von den einzelnen Gemeinden nicht erwartet werden, dass sie ihre konkreten Interessen zugunsten eines „abstrakten, bundesweiten Flächensparziels" zurückstellen (UBA 2003). Wenn allerdings konkrete Zahlen zum Grad der Zerschneidung vorliegen, können Handlungsziele festgelegt werden, um eine weitere Zunahme der Flächenzerschneidung einzugrenzen. Auch kann konkret überprüft werden, ob die von politischen Entscheidungsträgern angekündigten Handlungsziele tatsächlich eingehalten wurden. Eine räumlich differenzierte Betrachtung von Teilräumen kann aufzeigen wo Problemregionen liegen.

Nachdem im Auftrag des Hessischen Landesamtes für Umwelt und Geologie (HLUG) eine Studie durchgeführt wurde, die den Ist-Zustand der Landschaftszerschneidung in Hessen dokumentiert (Esswein et al. 2004a), werden mit der hier vorliegenden Untersuchung nun erstmals konkrete Zahlen zur Entwicklung der Landschaftszerschneidung in Hessen geliefert. Ziel dieser Untersuchung ist es, mit der gleichen Methode wie in Baden-Württemberg, Bayern und Schleswig-Holstein (u.a.), den Zerschneidungsgrad der Landschaft zu bewerten. Als Maßzahl diente die von Jaeger (2000) entwickelte *Effektive Maschenweite*.

Um eine räumlich differenzierte Bewertung zu ermöglichen, wurden insgesamt drei Ebenen im Bundesland Hessen betrachtet: Die drei hessischen Regierungsbezirke Darmstadt, Giessen und Kassel, die 26 Landkreise (21 Kreise und fünf kreisfreie Städte) und die 59 naturräumlichen Haupteinheiten Hessens. Gezeigt wird zunächst der aktuelle Zustand der Landschaftszerschneidung und nachfolgend in sechs Zeitschritten die historische Entwicklung der Siedlungsexpansion und des Verkehrswegenetzes von 1930 bis 2002. Erstmalig wird dann ein Vergleich der Entwicklung in den Bundesländern Hessen und Baden-Württemberg gezogen.

Die Ergebnisse, die in dieser Studie präsentiert werden, wurden im Rahmen eines Promotionsstipendiums der Deutschen Bundesstiftung Umwelt erarbeitet. (Roedenbeck 2005). Die vergleichenden Zahlen aus Baden-Württemberg (BW) resultieren aus einem Folgeprojekt zur Landschaftszerschneidung in BW, das von der Landesanstalt für Umweltschutz BW finanziert wurde (Esswein et al. 2005).

3.2 MATERIAL UND METHODEN

3.2.1 ZERSCHNEIDUNGSELEMENTE

Wenn die Zerschneidungsintensität einer realen Landschaft bewertet werden soll, muss im Vorfeld definiert werden, von welchen Landschaftselementen Zerschneidungswirkungen ausgehen. Für die vorliegende Studie wurden analog zu der Studie in Baden-Württemberg (Esswein et al. 2002) alle Strukturen ausgewählt, von denen Barrierewirkungen für die Wanderung von Tieren zu erwarten sind. Als Hindernis galten Straßen, Bahnlinien, Siedlungs- und Gewerbeflächen. Bei den Straßen wurden Autobahnen, Bundes-, Landes- und Kreisstrassen in die Analyse einbezogen. Gemeindeverbindungsstraßen wurden nur für den aktuellen Zustand berücksichtigt, nicht aber für die historische Trendanalyse, da ihre historische Entwicklung aus dem vorliegenden Kartenmaterial nicht abzuleiten war.

Viele Tierarten haben zusätzlich Probleme, Gewässer von größerer Breite zu überqueren. Demzufolge wurde auch die geogene Zerschneidung von Flüssen (breiter als 6m) und Seen berücksichtigt.

3.2.2 DATENVERARBEITUNG

Grundlage zur Berechnung der Zerschneidungssituation in Hessen waren digitale Datensätze des Amtlichen Topographisch-Kartographischen Informationssystems (ATKIS®). Für die Ist-Zustands-Analyse standen Daten der Realisierungsstufe 2 im Maßstab 1:25.000 (DLM 25/2, Stand 2002) zur Verfügung.

Datengrundlage für die historische Analyse war zunächst der digitale ATKIS-Datensatz der Realisierungsstufe 1 (DLM 25/1, Stand 1995). Für alle zeitliche früheren Jahrgänge lagen analoge Kartenblätter vor: Topographische Übersichtskarten von Hessen (TÜK 1:200.000) für die Jahre 1989, 1977 und 1966 sowie Kartenblätter des Deutschen Reiches im Maßstab 1:100 000 für das Jahr 1930. Die Straßen, Bahnlinien und Siedlungen wurden ausgehend von den 1995er ATKIS-Daten auf Grundlage der gescannten, georeferenzierten Kartenblätter schrittweise rückdigitalisiert. Nicht rückdigitalisiert wurden sämtliche Gewässer, da Flussbegradigungen, Verlandungen von Seen o.ä. nicht aus den topographischen Karten abzuleiten waren. Als Datengrundlage für die Gewässer diente der 2002er ATKIS-Datensatz, der über die untersuchte Zeitspanne konstant gehalten wurde.

Nachdem für alle Zieljahre die Zerschneidungselemente digital vorlagen, wurden mit der GIS-Software ArcView (Version 3.2) und ArcInfo (Version 8.0.1) alle zerschneidungsrelevanten flächen- und linienhaften Elemente räumlich überlagert. Ziel der Methode ist es, ein Flächenmosaik zu generieren, das die Landesfläche von Hessen bedeckt. Das fertige Flächenmosaik besteht aus aneinander angrenzenden Polygonen (Teilflächen). Die polygonbildenden Grenzlinien sind die jeweiligen Zerschneidungselemente (z.B. Bahnlinien, Strassen oder Ränder der Ortslagen).

Im Zuge der räumlichen Überlagerung wurden alle Polygone attributiert, d.h., ihnen wurde eine Flächeneigenschaft zugewiesen. Dieses Vorgehen ermöglicht es, Siedlungen und flächenhafte Gewässer aus dem Flächenmosaik auszuschneiden. Übrig bleibt als Endergebnis ein Polygonnetz, das ausschließlich aus unzerschnittenen Freiräumen besteht. Diese Teilflächen gehen in die Berechnung der effektiven Maschenweite ein.

3.2.3 DIE EFFEKTIVE MASCHENWEITE

Die *effektive Maschenweite* (m_{eff}) ist ein Ausdruck für die Möglichkeit, dass sich zwei Tiere, die zufällig und unabhängig voneinander in einem Gebiet ausgesetzt werden, begegnen können (Jaeger 2000). Je mehr Barrieren in die Landschaft eingefügt werden, umso geringer wird die Begegnungswahrscheinlichkeit. Für die Berechnung von m_{eff} werden zwei Angaben benötigt:

(1) Die Gesamt-Flächengröße F_g eines untersuchten Gebietes und

(2) die Flächengröße F_i jedes i-ten unzerschnittenen Teilraums in dem untersuchten Gebiet.

$$m_{eff} = F_g * \sum_{i=1}^{n} \left(\frac{F_i}{F_g} \right)^2 = \frac{\sum F_i^2}{F_g}$$

Je größer der Wert der effektiven Maschenweite eines betrachteten Gebiets ist, desto geringer ist sein Zerschneidungsgrad. Die m_{eff} kann maximal den Wert der Fläche des untersuchten Gebietes (bzw. Teilraumes) annehmen, wenn keinerlei trennende Elemente vorkommen. Im Extremfall nimmt sie den Wert 0 an, wenn alle Flächen vollständig versiegelt, bzw. mit Trennelementen bedeckt sind. Um die effektive Maschenweite einsetzen zu können, ist die Vorgabe eines oder mehrerer Bezugsräume notwendig. Nach Esswein et al. (2002) kann die Bezugsraumgrenze nach zwei verschiedenen Verfahren festgelegt werden (**Fig. 3.1**).

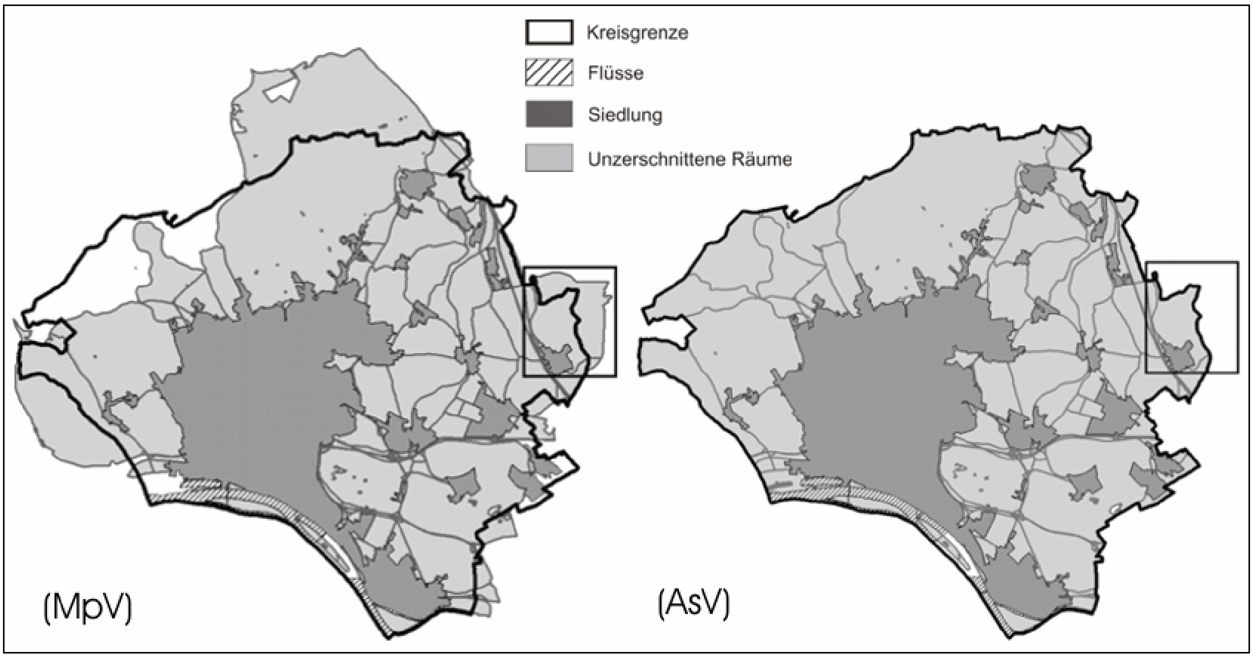


***Fig. 3.1**: Mittelpunkt- (MpV) und Ausschneideverfahren (AsV) im Vergleich - am Beispiel der kreisfreien Stadt Wiesbaden (Hessen). Beim MpV (links) gehen alle unzerschnittenen Räume in die Berechnung ein, die ihren Mittelpunkt im Bezugsraum haben. Beim AsV (rechts) wirkt die Bezugsraumgrenze als zusätzliche, künstliche Barriere.*

(1) Beim Mittelpunktverfahren (MpV) werden alle unzerschnittenen Freiräume dem Bezugsraum zugeordnet, die ihren Mittelpunkt im Bezugsraum haben. Dieses Verfahren ermöglicht keine historische Trendanalyse, weil die Bezugsraumgrenzen im Zeitverlauf ständig variieren.

(2) Beim Ausschneideverfahren (AsV) gilt die Bezugsraumgrenze als zusätzliche, künstliche Barriere. Alle Teilflächen in Grenzlage werden nach Vorgabe der Grenzlinie zerteilt und es gehen nur die Teilflächen in die Berechnung ein, die exakt im Bezugsraum liegen. Mit diesem Verfahren ist eine historische Analyse möglich, weil die Bezugsraumgrenzen konstant gehalten werden. Die hier dargestellten Ergebnisse wurden alle mit dem Ausschneideverfahren ermittelt.

3.3 ERGEBNISSE

3.3.1 LANDESWEITER ÜBERBLICK - HESSEN IM VERGLEICH ZU BADEN-WÜRTTEMBERG

Das Bundesland Hessen ist 2002 mit Einbezug der Gemeindestraßen in 15.260 einzelne Teilräume zerschnitten. 17 dieser Räume (5,3 % der Landesfläche) sind größer als 50 km², ein einzelner Raum ist mit 105,46 km² (0,5% der Landesfläche) größer als 100 km². Die effektive Maschenweite für Gesamthessen beträgt 15,63 km². Der Definition zur effektiven Maschenweite von Jaeger (2000) folgend bedeutet dies: Wenn alle unzerschnittenen Räume nach der Art eines gleichmäßigen Rasters über die Landesfläche verteilt wären, hätten die Rasterzellen alle eine Größe von 3,953 km x 3,953 km = 15,63 km². In diesem Fall entspräche die effektive Maschenweite der Durchschnittsgröße der unzerschnittenen Räume.

Ohne Gemeindestraßen ist Hessen in 10.458 Teilflächen zerteilt. Die Effektive Maschenweite beträgt dann 16,59 km². Der geringe Unterschied der Werte zeigt, dass Hessen insgesamt relativ wenig Gemeindestraßen aufweist. In Baden-Württemberg variieren die Werte vergleichsweise stärker. Die aktuellen Ergebnisse (2002) messen für die Geometrie mit Gemeindestraßen einen Wert von 13,01 km², ohne Gemeindestraßen steigt m_{eff} auf 19,58 km². Ein Vergleich beider Bundesländer ist somit davon abhängig, ob Gemeindestraßen einbezogen werden oder nicht.

Betrachtet man die historische Entwicklung der Ortslagen und des Verkehrswegenetzes ohne Gemeindestraßen, so hat die Landschaftszerschneidung in Hessen seit 1930 kontinuierlich zugenommen (**Fig. 3.2**, **Tab. 3.1**). Von 1930 bis 2002 sinkt der Wert der effektiven Maschenweite von 22,10 auf 16,59 km². Dies entspricht einer Abnahme um rund 25%. Der größte Sprung ist zwischen 1930 und 1966 zu verzeichnen. Hier sinkt die effektive Maschenweite um 4,11 km². Die Anzahl der Räume größer 50 km² sank von 1930 bis 2002 von 31 auf 18. Das bedeutet einen Rückgang von rund elf auf rund sechs Prozent der Landesfläche. 1930 existierten noch vier Flächen größer als 100 km² in Hessen. Bereits 1966 blieb nur eine Fläche hiervon übrig, die rund 0,5 % der Landesfläche bedeckt und im Wesentlichen bis 2002 bestehen bleibt. Der seit 1966 größte unzerschnittene Raum Hessens liegt im Rheingau-Taunus-Kreis, westlich von Wiesbaden.

Ein Vergleich zwischen Hessen und Baden-Württemberg ohne Gemeindestraßen zeigt, das Hessen insgesamt stärker zerschnitten war und ist (**Fig. 3.2**, **Tab. 3.1**). Allerdings hat sich mit einer Reduktion der effektiven Maschenweite um 38%, von 31,60 km² auf 19,58 km², im betrachteten Zeitraum die Situation in Baden-Württemberg deutlich schlechter entwickelt als in Hessen. Zwar gibt es in Baden-Württemberg insgesamt jeweils mehr Räume größer 50 bzw. 100 km², jedoch sind die Rückgänge in Anzahl und Prozent gravierender. Die Räume über 50 km² sind bis 2004 von 83 auf 39 gesunken, ihr Anteil an der Landesfläche reduzierte sich somit um ca. 10% von 18,7 auf 8,8. Seit 1977 gibt es in Baden-Württemberg lediglich noch acht Flächen über 100 km², die noch 3,1 % der Landesfläche bedecken. Im Vergleich zu 1930 bedeutet dies einen Rückgang um 3,5 %, von 6,6 auf 3,1. Seit 1930 gingen damit neun unzerschnittene Räume über 100 km² verloren (Esswein et al. 2005).

Tab. 3.1: *Entwicklung der Landschaftszerschneidung in Baden-Württemberg, Hessen, und in den drei hessischen Regierungsbezirken von 1930-2002. Für das Jahr 2002 wurde der Zerschneidungsgrad einmal unter Einbezug der Gemeindestraßen (m.G.) und einmal ohne Gemeindestraßen (o.G.) berechnet.*

Regierungspräsidium (bzw. Bundesland)	Fläche [km²]	Effektive Maschenweite [km²]							Veränderung gegenüber 1930 Ebene o.G.
		1930 o.G.	1966 o.G.	1977 o.G.	1989 o.G.	1995 o.G.	2002 o.G.	2002 m.G.	
Baden-Württemberg	**35750**	**31,60**	**24,26**	**22,14**	**20,51**	**20,24**	**19,58**	**13,01**	**38,04%**
Hessen	**21116**	**22,10**	**17,99**	**17,25**	**16,82**	**16,74**	**16,59**	**15,63**	**24,93%**
Darmstadt	7445	21,35	16,43	15,53	15,31	15,19	15,04	14,20	29,56%
Giessen	5381	16,60	15,12	14,38	14,24	14,23	14,12	13,27	14,94%
Kassel	8291	24,23	19,52	18,97	18,20	18,10	17,94	16,60	25,96%

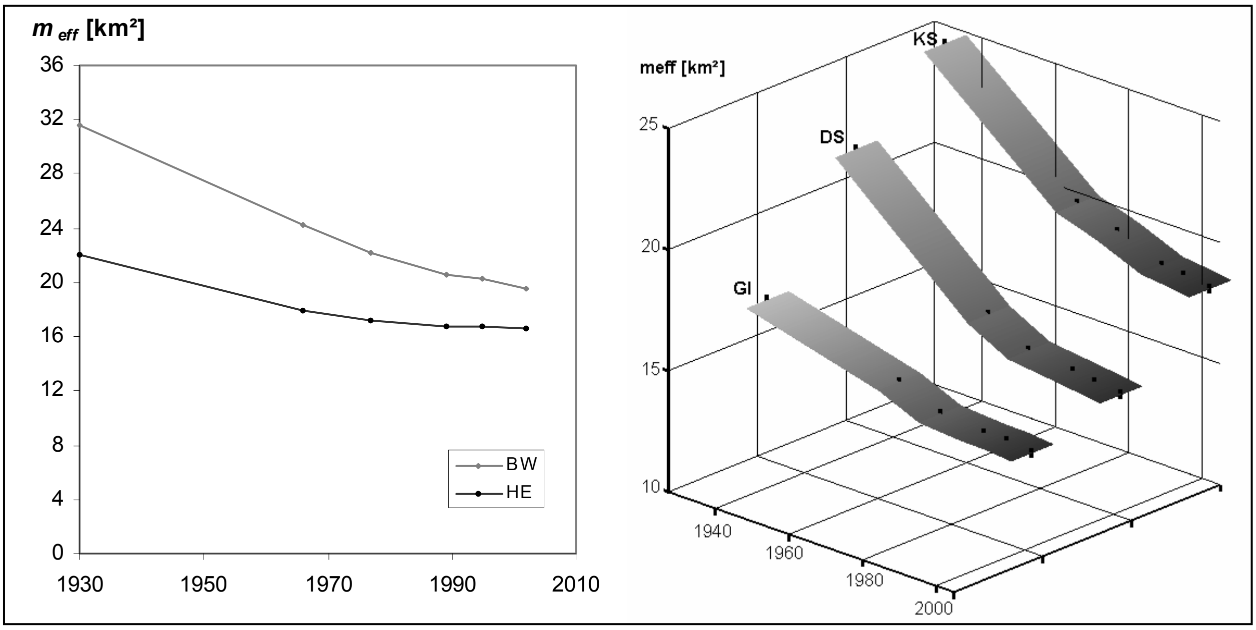


Fig. 3.2 (links): *Vergleichende Entwicklung der Landschaftszerschneidung in Hessen und Baden-Württemberg. Eine Abnahme der effektiven Maschenweite (m_{eff}) bedeutet eine Zunahme der Zerschneidung. Eine Zerschneidungswirkung wurde folgenden Landschaftsstrukturen zugeordnet: Autobahnen, Bundes-, Landes- und Kreisstraßen, Bahnlinien, Seen, Gewässer (>6m Breite) und Siedlungen.*

Fig. 3.3 (rechts)*: Entwicklung der Landschaftszerschneidung in den drei hessischen Regierungsbezirken Kassel (KS), Darmstadt (DS) und Giessen (GI).*

3.3.2 DIE HESSISCHEN REGIERUNGSBEZIRKE

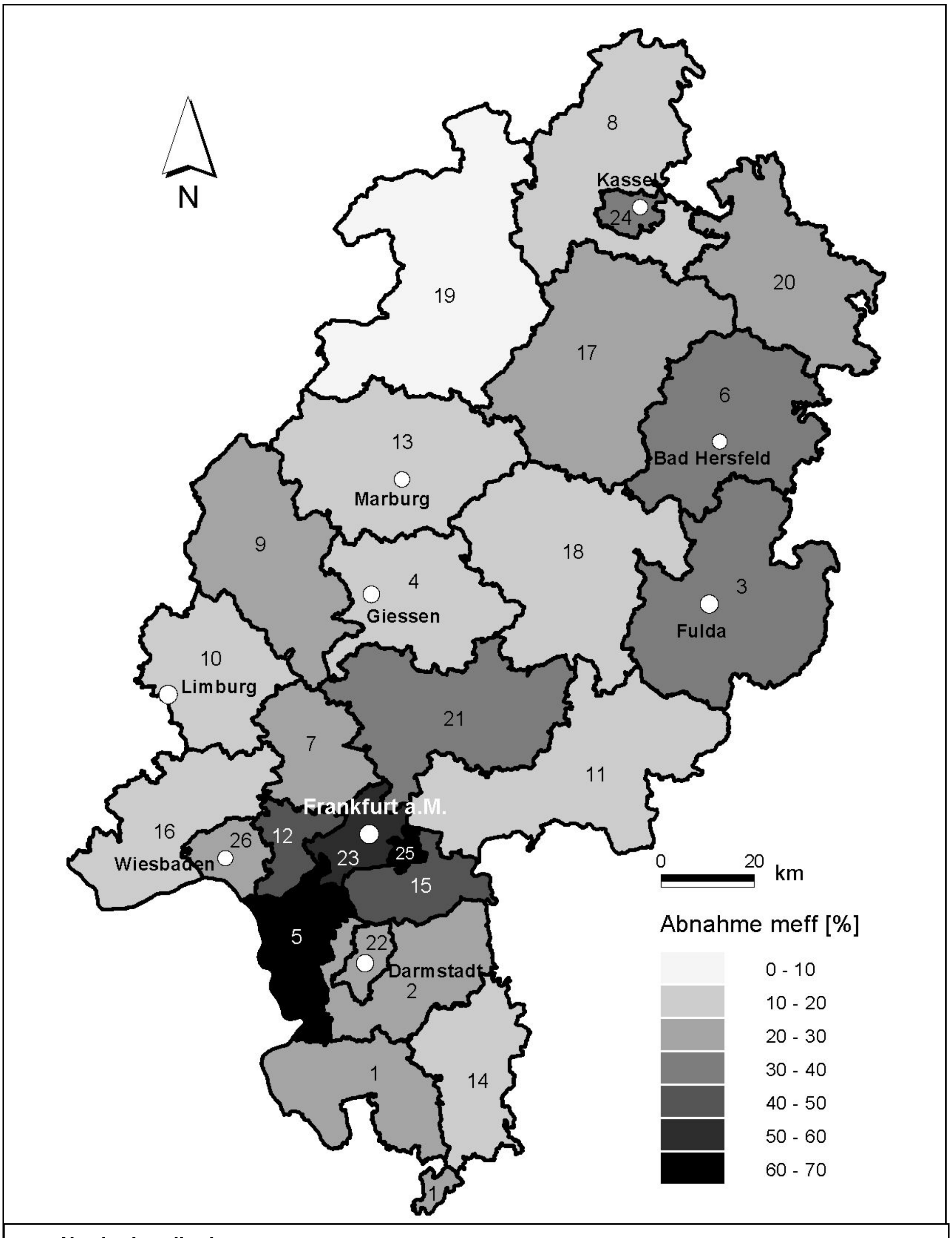
Betrachtet man im Jahr 2002 die drei hessischen Regierungsbezirke unter Einbezug der Gemeindestraßen, so ist Giessen am stärksten zerschnitten (m_{eff}=13,27), gefolgt von Darmstadt (m_{eff}=14,20) und Kassel (m_{eff}=16,60). Kassel ist als einziger Bezirk geringer zerschnitten als das gesamte Bundesland (**Fig. 3.3**, **Tab. 3.1**).

Die historische Trendanalyse ohne Gemeindestraßen verdeutlicht, dass diese Hierarchie seit 1930 über rund 70 Jahre bis heute beibehalten wird. In allen drei Regierungsbezirken treten die größten Veränderungen zwischen 1930 und 1966 auf. Obwohl Giessen insgesamt immer am stärksten zerschnitten war und ist, sind Darmstadt und Kassel von stärkeren Veränderungen im Zeitablauf betroffen. In Darmstadt ist bis 2002 eine Abnahme der effektiven Maschenweite um rund 30% zu verzeichnen, in Kassel um 26%, in Giessen dagegen „nur“ um 15%.

3.3.3 DIE HESSISCHEN LANDKREISE

Im Jahr 2002 beträgt der Mittelwert der effektiven Maschenweite aller Landkreise mit Gemeindestraßen 11,02 km², der Median liegt bei 10,61 km². Insgesamt variieren die Werte für die Landkreise extrem zwischen der stark zerschnittenen Stadt Offenbach (m_{eff}=1,12) und dem am wenigsten zerschnittenen Landkreis Rheingau-Taunus (m_{eff}=22,74). Es ist einleuchtend, dass die fünf Stadtkreise auf den obersten Rängen der Zerschneidungsstärke rangieren. Einzige Ausnahme ist der Main-Taunus-Kreis auf Rang drei (m_{eff}=3,63). 19 von 26 Landkreisen sind stärker zerschnitten als der Wert für Gesamthessen (m_{eff}=15,50).

Die seit 1930 verlaufende Verdichtung des Verkehrswegenetzes und die Siedlungsexpansion in Hessen kann durch eine Trend-Betrachtung in den hessischen Landkreisen räumlich präzisiert werden (**Tab. 3.2**). Ohne Ausnahme nimmt in allen Landkreisen und kreisfreien Städten die Flächenzerschneidung seit 1930 kontinuierlich zu. Fünf von 26 Landkreisen zeigen prozentuale Veränderungen des Zerschneidungsgrades von über 40%. Dies sind Main-Taunus (49%), Offenbach (48%) und Stadt Frankfurt (54%) sowie Gross-Gerau und Stadt Offenbach. Die beiden letztgenannten verzeichnen die negativsten Trends innerhalb von 70 Jahren mit Rückgängen der effektiven Maschenweite von mehr als 60%. Relativ geringe Veränderungen weisen die Landkreise Waldeck-Frankenberg (8,23%), Odenwald (11,07%) und Marburg-Biedenkopf (11,73%) auf (**Fig. 3.4**). Die Entwicklung spielt sich in verschiedenen Zeitfenstern ab. Es lassen sich grob drei Typen unterscheiden: Typ 1 zeichnet sich durch einen starken Abfall der effektiven Maschenweite bis 1966 aus (Hersfeld-Rotenburg, Werra-Meissner, Gross-Gerau und die kreisfreien Städte Frankfurt am Main und Kassel). Charakteristisch für Typ 2 sind starke Veränderungen im Zeitfenster bis 1977 (Bergstrasse, Vogelsberg, Lahn-Dill, Hochtaunus, Wetterau und Waldeck-Frankenberg, sowie die Städte Offenbach und Wiesbaden). In Typ 3 lassen sich alle übrigen Landkreise zusammenfassen, die einen kontinuierlichen Abfall der effektiven Maschenweite zeigen. Die stärksten Veränderungen im Zeitverlauf zeigen in dieser Typgruppe die Landkreise Fulda und Main-Taunus.



Nr. der Landkreise:

1: Bergstrasse	8: Kassel	14: Odenwald	20: Werra-Meissner
2: Darmstadt-Dieburg	9: Lahn-Dill	15: Offenbach	21: Wetterau
3: Fulda	10: Limburg-Weilburg	16: Rheingau-Taunus	22: St. Darmstadt
4: Giessen	11: Main-Kinzig	17: Schwalm-Eder	23: St. Frankfurt
5: Gross-Gerau	12: Main-Taunus	18: Vogelsberg	24: St. Kassel
6: Hersfeld-Rotenburg	13: Marburg-Biedenkopf	19: Waldeck-Frankenberg	25: St. Offenbach
7: Hochtaunus			26: St. Wiesbaden

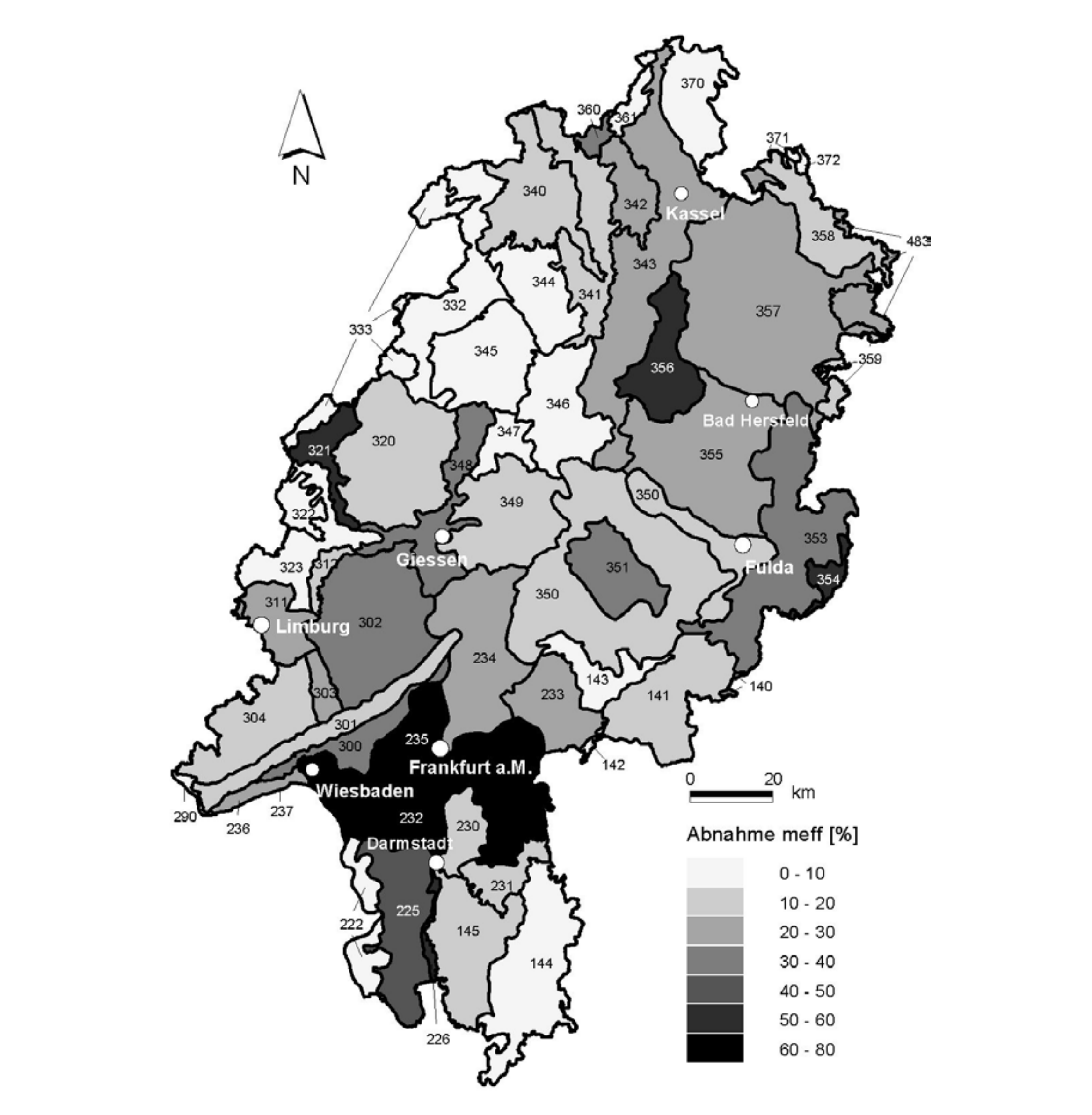
Fig. 3.4: *Prozentuale Abnahme der effektiven Maschenweite (m_{eff}) in den hessischen Landkreisen von 1930 bis 2002 (ohne Gemeindestraßen). Datengrundlage: ATKIS-Basis-DLM 1 (Stand 2002, Maßstab 1:25.000) und Karten des Deutschen Reiches um 1930 (Staatsbibliothek zu Berlin).*

Tab. 3.2: *Entwicklung der Landschaftszerschneidung in den hessischen Landkreisen und kreisfreien Städten von 1930-2002. Für das Jahr 2002 wurde der Zerschneidungsgrad einmal unter Einbezug der Gemeindestraßen (m.G.) und einmal ohne Gemeindestraßen (o.G.) berechnet.*

Landkreis	Fläche [km²]	Effektive Maschenweite [km²]							Veränderungen gg. 1930 [%]
		1930	1966	1977	1989	1995	2002	2002	
		o.G.	o.G.	o.G.	o.G.	o.G.	o.G.	m.G.	Ebene o.G.
Bergstrasse	719,48	22,67	18,59	16,88	16,66	16,51	16,19	12,64	28,58
Darmstadt-Dieburg	658,46	11,63	10,13	9,52	9,35	9,26	9,16	8,46	21,24
Fulda	1380,66	18,19	14,40	13,28	12,83	12,60	12,30	11,17	32,38
Giessen	854,58	10,79	9,79	9,52	9,02	8,93	8,84	8,6	18,07
Gross-Gerau	453,05	25,64	10,08	10,51	10,30	10,18	10,06	8,46	60,76
Hersfeld-Rotenburg	1097,71	34,37	25,66	24,81	23,39	23,25	23,09	22,58	32,82
Hochtaunus	481,91	16,85	14,06	12,95	12,89	12,87	12,87	11,74	23,62
Kassel	1293,37	23,63	20,85	20,54	20,15	20,09	19,96	15,20	15,53
Lahn-Dill	1066,19	16,80	14,45	13,09	13,04	13,12	12,96	11,34	22,86
Limburg-Weilburg	738,50	7,89	7,08	6,91	6,86	6,76	6,63	6,43	15,97
Main-Kinzig	1397,38	21,29	18,42	17,81	17,57	17,48	17,34	17,17	18,55
Main-Taunus	222,48	7,36	4,94	4,28	3,98	3,85	3,76	3,63	48,91
Marburg-Biedenkopf	1262,49	18,50	16,62	16,64	16,53	16,50	16,33	15,28	11,73
Odenwald	624,01	19,33	18,19	17,44	17,37	17,33	17,19	18,87	11,07
Offenbach	356,06	12,44	8,21	6,99	6,79	6,67	6,53	5,92	47,51
Rheingau-Taunus	811,47	25,83	24,08	23,67	23,38	23,18	23,01	22,74	10,92
Schwalm-Eder	1539,05	14,33	11,97	11,69	10,67	10,74	10,66	10,05	25,61
Vogelsberg	1458,80	18,52	17,60	16,18	16,13	16,12	16,15	15,85	12,80
Waldeck-Frankenberg	1848,82	20,17	19,08	18,90	18,66	18,64	18,51	18,04	8,23
Werra-Meissner	1024,73	23,05	18,16	18,04	17,93	17,63	17,64	17,14	23,47
Wetterau	1100,73	15,20	11,52	10,44	10,51	10,43	10,35	9,70	31,91
Stadt Darmstadt	122,22	8,00	6,41	6,16	5,98	5,96	5,89	5,48	26,38
Stadt Frankfurt a. M.	248,41	4,95	2,29	2,30	2,40	2,31	2,26	1,26	54,34
Stadt Kassel	106,75	3,34	2,47	2,43	2,41	2,36	2,33	2,27	30,24
Stadt Offenbach	44,83	3,69	1,82	1,39	1,35	1,34	1,31	1,12	64,50
Stadt Wiesbaden	203,89	7,84	6,77	5,99	5,85	5,76	5,65	5,39	27,93
Mittelwert		15,86	12,83	12,24	12,00	11,92	11,81	11,02	
Median		16,83	13,02	12,32	11,75	11,67	11,48	10,61	

3.3.4 DIE HESSISCHEN NATURRÄUME

Im Jahr 2002 beträgt der Mittelwert der effektiven Maschenweite aller Naturräume mit Gemeindestraßen 9,78 km², der Median liegt bei 7,96 km². Die drei größten unzerschnitten Naturräume sind der Büdinger Wald (m_{eff}=25,66), Solling (m_{eff}=27,16) und der Sandsteinspessart (m_{eff}=27,24). 49 von 59 Naturräumen sind stärker zerschnitten als der Wert für Gesamthessen (m_{eff}=15,50). Die historische Entwicklung der Landschaftszerschneidung ohne Gemeindestraßen verläuft in den Naturräumen unterschiedlich stark (**Tab. 3.3**). Nur geringe Abnahmen der effektiven Maschenweite im Zeitverlauf (unter 5% des Wertes von 1930) zeigen die Naturräume Oberwälder Land (2,05%), Hochsauerland (3,38%), Kellerwald (4,18%), Sollingvorland (4,23%) und Burgwald (4,39 %). Besonders drastische Entwicklungen - mit prozentualen Abnahmen der effektiven Maschenweite von über 50% gegenüber dem Wert von 1930 - zeigen dagegen die Naturräume Untermainebene (70,22%), Main-Taunusvorland (64,96%), Bergstrasse (57,23%), Hohe Rhön (56, 15%), Dilltal (54,08%) und Knüll-Hochland (53,12%) (**Fig. 3.5**).



Naturräumliche Haupteinheiten:

140 Südrhön
141 Sandsteinspessart
142 Vorderer Spessart
143 Büdinger Wald
144 Sandsteinodenwald
145 Vorderer Odenwald
222 Nördliche Oberrheinniederung
225 Hessische Rheinebene
226 Bergstrasse
230 Messeler Hügelland
231 Reinheimer Hügelland
232 Untermainebene
233 Büdingen-Meerholzer Hügelland
234 Wetterau
235 Main-Taunusvorland
236 Rheingau
237 Ingelheimer Rheinebene
290 Oberes Mittelrheintal
300 Vortaunus
301 Hoher Taunus
302 Östlicher Hintertaunus
303 Idsteiner Senke
304 Westlicher Hintertaunus
311 Limburger Becken
312 Weilburger Lahntal
320 Gladenbacher Bergland
321 Dilltal
322 Hoher Westerwald
323 Oberwesterwald
324 Niederwesterwald
331 Siegerland
332 Hochsauerländer Gebirgsrand
333 Hochsauerland
340 Waldecker Tafel
341 Ostwaldecker Randsenken
342 Habichtswälder Bergland
343 Westhessische Senke
344 Kellerwald
345 Burgwald
346 Oberhessische Schwelle
347 Amöneburger Becken
348 Marburg-Gießener Lahntal
349 Vorderer Vogelsberg
350 Unterer Vogelsberg
351 Hoher Vogelsberg (mit Oberwald)
352 Fuldaer Senke
353 Vorder- und Kuppenrhön
354 Rhön
355 Fulda-Haune-Tafelland
356 Knüll-Hochland
357 Fulda-Werra-Bergland
358 Unteres Werratal
359 Salzunger Werrabergland
360 Warburger Börde
370 Solling, Bramwald, Reinhardswald
371 Sollingvorland
372 Lein-Ilme-Senke
483 NW Randpl. Thüringer Becken

***Fig. 3.5**: Prozentuale Abnahme der effektiven Maschenweite in den hessischen Naturräumen von 1930 bis 2002. Datengrundlage: ATKIS-Basis-DLM 1 (Stand 2002, Maßstab 1:25.000) und Karten des Deutschen Reiches um 1930 (Staatsbibliothek zu Berlin).*

Tab. 3.3: *Entwicklung der Landschaftszerschneidung in den hessischen Naturräumen von 1930-2002. Für das Jahr 2002 wurde der Zerschneidungsgrad einmal mit und einmal ohne Gemeindestraßen berechnet (m.G. bzw. o.G.).*

Naturraum	Fläche [km²]	Effektive Maschenweite [km²]							Veränd. gg. 1930 [%]
		1930	1966	1977	1989	1995	2002	2002	
		o.G.	o.G.	o.G.	o.G.	o.G.	o.G.	m.G.	Ebene o.G.
Amöneburger Becken	136,4	4,39	4,18	4,3	4,18	4,17	4,09	3,64	6,83
Bergstrasse	38,8	4,7	2,62	2,54	2,07	2,03	2,01	1,78	57,23
Büdingen-Meerholzer Hügelland	323,3	9,84	8,28	7,39	7,43	7,28	7,22	7,13	26,63
Büdinger Wald	202,4	27,48	26,55	25,95	25,94	25,93	25,76	25,66	6,26
Burgwald	493,5	24,38	23,3	23,19	23,44	23,43	23,31	22,1	4,39
Dilltal	169,6	15,2	8,02	7,3	7,13	7,12	6,98	6,41	54,08
Fulda-Haune-Tafelland	911,9	25,32	21,5	20,65	19,58	19,62	19,47	19,12	23,10
Fulda-Werra-Bergland	1561,2	33,93	28,37	27,97	26,08	26,04	25,95	25,4	23,52
Fuldaer Senke	292,3	5,54	5,15	4,94	4,87	4,86	4,7	4,48	15,16
Gladenbacher Bergland	779,8	18,72	16,58	16,06	15,96	15,93	15,82	14,43	15,49
Habichtswälder Bergland	205,1	14,26	14,11	11,55	11,33	11,32	11,2	10,68	21,46
Hessische Rheinebene	492,0	17,87	10,87	10,08	9,79	9,63	9,53	8,94	46,67
Hochsauerland (Rothaargebirge)	213,5	18,32	17,99	17,85	17,78	17,78	17,70	14,04	3,38
Hohe Rhoen	79,5	18,17	8,44	8,2	8,2	8,19	7,97	7,96	56,14
Hoher Taunus	313,3	22,31	19,57	19,19	19,13	19,08	18,94	17,32	15,11
Hoher Vogelsberg (mit Oberwald)	325,6	22,28	21,69	14,33	14,74	14,73	14,77	14,65	33,71
Hoher Westerwald	113,8	6,63	6	5,97	6,12	6,12	6,13	6,06	7,54
Idsteiner Senke	84,0	5,69	4,6	4,5	4,17	4,13	4,04	3,95	29,00
Ingelheimer Rheinebene	19,8	0,26	0,32	0,31	0,31	0,31	0,29	0,22	-11,54
Kellerwald	346,0	25,11	24,57	24,57	24,09	24,1	24,06	24,19	4,18
Knüll-Hochland	354,9	28,54	14,55	14,51	13,36	13,47	13,38	11,01	53,12
Lein-Ilme-Senke	13,8	2,54	2,51	2,51	2,5	2,5	2,39	1,65	5,91
Limburger Becken	223,2	4,7	4,04	3,83	3,77	3,76	3,61	3,25	23,19
Main-Taunusvorland	319,7	5,68	3,3	2,53	2,2	2,11	1,99	1,79	64,96
Marburg-Gießener Lahntal	390,8	4,82	4,2	3,64	3,22	3,21	3,12	2,9	35,27
Messeler Hügelland	171,4	14,46	13,76	12,9	12,87	12,83	12,78	12,64	11,62
Niederwesterwald	7,4	2,71	2,15	2,15	2,15	2,15	2,03	2,01	25,09
Nördliche Oberrheinniederung	194,1	11,17	10,86	10,63	10,53	10,53	10,4	7,69	6,89
NW Randplatten d. Thür. Beckens	133,7	17,47	13,43	13,43	13,43	13,43	13,42	13,41	23,18
Oberes Mittelrheintal	21,3	4,78	4,61	4,48	4,48	4,41	4,41	4,41	7,74
Oberhessische Schwelle	454,3	14,86	13,68	13,63	13,62	13,57	13,48	11,84	9,29
Oberwälder Land	86,8	10,73	10,58	10,57	10,56	10,55	10,51	10,49	2,05
Oberwesterwald	341,3	9,97	9,22	8,93	8,9	9,21	9,03	8,09	9,43
Östlicher Hintertaunus	824,3	19,7	13,26	12,87	12,88	12,85	12,49	12,06	36,60
Ostsauerländer Gebirgswand	534,3	20,56	19,52	19,6	19,47	19,43	19,31	19,05	6,08
Ostwaldecker Randsenken	475,4	9,4	9,09	8,4	8,14	8,13	8,06	7,99	14,26
Reinheimer Hügelland	163,8	6,89	5,69	5,65	5,64	5,64	5,57	5,55	19,16
Rheingau	73,8	7,68	6,71	6,22	6,29	5,95	5,7	5,47	25,78
Salzunger Werrabergland	64,8	8,24	6,82	6,68	6,68	6,67	6,67	5,628	19,05
Sandsteinodenwald	672,4	21,19	20,07	19,36	19,3	19,24	19,11	20,65	9,82
Sandsteinspessart	529,7	31,3	27,47	27,27	27,14	27,06	27,05	27,24	13,58
Solling, Bramwald, Rheinhardswald	341,7	49,11	45,12	45	44,75	44,66	44,55	27,16	9,29
Sollingvorland	8,8	3,78	3,78	3,78	3,63	3,63	3,62	2,988	4,23
Südrhön	7,8	4,17	4,17	4,17	2,63	2,63	2,63	2,63	36,93
Unterer Vogelsberg	1250,1	14,84	13,64	13,11	13,06	13,04	13,05	12,84	12,06
Unteres Werraland	317,7	7,42	6,42	6,35	6,3	6,02	6,06	6,02	18,33
Untermainebene	1034,9	21,12	7,55	6,91	6,6	6,46	6,29	5,43	70,22
Vorder- und Kuppenrhön	883,4	15	11,8	11,08	10,77	10,5	10,1	9,01	32,67
Vorderer Odenwald	506,1	20,68	19,22	17,66	17,59	17,38	17,1	12,96	17,31
Vorderer Spessart	14,5	6,25	5,8	5,76	5,76	5,76	5,76	5,76	7,84
Vorderer Vogelsberg	571,6	11,75	11,28	11,14	10,62	10,48	10,41	9,78	11,40
Vortaunus	215,6	7,12	5,13	4,69	4,66	4,6	4,61	4,45	35,25
Waldecker Tafel	514,7	15,86	14,36	14,38	14,1	14,08	13,94	13,66	12,11
Warburger Börde	51,0	6,60	6,50	4,54	4,53	4,53	4,49	4,48	31,97
Weilburger Lahntal	63,3	3,1	2,5	2,44	2,42	2,54	2,49	2,45	19,68
Westhessische Senke	1034,4	5,89	4,74	4,68	4,33	4,29	4,2	4,12	28,69
Westlicher Hintertaunus	474,7	16,48	15,07	14,8	14,34	14,3	14,31	14,08	13,17
Wetterau	672,8	8,96	7,42	6,88	6,77	6,57	6,54	6,21	27,01
Mittelwert		13,49	11,4	10,92	10,72	10,7	10,59	9,78	
Median		11,17	9,09	8,4	8,2	8,19	8,06	7,96	

Analog zu der Entwicklung in den Kreisen sind auch hier hauptsächlich drei Entwicklungstypen zu unterscheiden. In vielen Naturräumen finden die stärksten Veränderungen in der Zeit bis 1966 statt (Typ 1), so z.B. im Knüll-Hochland, Untermainebene, Hohe Rhön und Hessische Rheinebene. Andere Naturräume fallen unter den Typ 2 mit teilweise starken Veränderungen bis 1977, so z.B. die Naturräume Hoher Vogelsberg, Dilltal, Habichtswälder Bergland und Vortaunus. In anderen Naturräumen (Typ 3) hat die Flächenzerschneidung zwar langsam aber kontinuierlich zugenommen (z.B. Limburger Becken und Fuldaer Senke).

In wenigen Naturräumen ist auch eine „Entschneidung" spürbar. So hat beispielsweise im Weilburger Lahntal die effektive Maschenweite von 1989 bis 1995 zugenommen, was einer Abnahme der Flächenzerschneidung entspricht. Ähnliche Entwicklungen zeigen Oberwesterwald (zwischen 1989 und 1995), Unteres Werraland (1995-2002), Hoher Westerwald (1977-1989) und Amöneburger Becken (1966-1977). In allen Fällen kommen Zunahmen der effektiven Maschenweite dadurch zustande, dass ehemals betriebene Bahnlinien im Zeitverlauf stillgelegt wurden und damit im Rahmen dieser Untersuchung nicht mehr als zerschneidungsrelevant gewertet werden.

3.4 DISKUSSION

Ein Vergleich der aktuellen Situation in Hessen mit dem Ist-Zustand anderer Bundesländer ist eingeschränkt möglich, sofern die Zerschneidungselemente in allen Untersuchungen einheitlich gewählt wurden (Esswein et al. 2004b). Auf der Ebene mit Gemeindestraßen können als Vergleich die Ergebnisse aus Baden-Württemberg (Esswein et al 2002), Nordrhein-Westfalen (Baumann und Hinterlang 2001) und Sachsen (LFUG 2002) herangezogen werden. Es zeigt sich, dass Hessen an dritter Stelle rangiert, hinter dem stark zerschnittenen NRW (m_{eff}= 9,51) und Baden-Württemberg (m_{eff} = 13,66), und vor dem am wenigsten zerschnittenen Sachsen (m_{eff}= 18,2). Auf der Ebene ohne Gemeindestraßen bieten sich zwei Bundesländer zum Vergleich an. Dabei ist Hessen (m_{eff}= 16,59) im Vergleich zu Baden-Württemberg (m_{eff}= 20,24) und Bayern (m_{eff}= 35,25) am stärksten zerschnitten.

Die historische Trendanalyse zeigt, dass die Landschaftszerschneidung in Hessen seit 1930 ununterbrochen zugenommen hat. Entschneidungen sind meist nur dann spürbar, wenn Bahnlinien im Laufe der Zeit aus dem Betrieb genommen wurden. Die historische Besiedlung der Region folgte den Flusstälern und mied die Höhenlagen. Dieses Siedlungsverhalten prägt Hessen bis heute, und so lässt sich in der Karte der unzerschnittenen Räume generell die Höhenlage der hessischen Naturräume wieder erkennen. Am stärksten zerschnitten sind Main- und Rheinebene im Ballungsgebiet zwischen Wiesbaden, Frankfurt und Darmstadt. Auch das Lahntal zwischen Giessen, Marburg und Limburg lässt sich als Besiedlungsachse wieder erkennen, ebenso wie das Schwalm-Ohm-Tiefland. Die Wetterau als Verbindungsachse zwischen Lahntal und Rheinebene, sowie das Amöneburger Becken fallen als Tieflandgebiete mit fruchtbaren Böden auf, die wegen ihrer landwirtschaftlichen Nutzbarkeit früh urbanisiert und später stark zersiedelt wurden.

Die größeren unzerschnittenen Räume liegen in den hessischen Mittelgebirgen Taunus, Hessisch-Fränkisches Bergland, Ausläufern des Hochsauerlandes, Westerwald, Kellerwald, Burgwald und Solling. Diese Hochlagen waren historisch wegen ihrer erschwerten landwirtschaftlichen Nutzbarkeit spärlich besiedelt, und sind bis heute in der Karte der unzerschnittenen Räume als größere Freiräume erkennbar. Deutlich wird, dass die Hochlagen in der Nähe der Ballungsgebiete stärker zersiedelt sind. Zum einen wegen ihrer Nutzbarkeit als Naherholungsgebiet (z.B. Spessart für das Rhein-Main-Gebiet), zum anderen, weil sie in der Verbindungsachse zwischen Siedlungsschwerpunkten liegen (z.B. Taunus zwischen Frankfurt/Wiesbaden und Limburger Lahntal). Im Vergleich weniger beansprucht sind die peripheren Mittelgebirge bei Kassel (Werrabergland) entlang der ehemaligen Grenze zur DDR.

3.5 Schlussfolgerungen und Ausblick

Die vorliegende Untersuchung liefert nun auch für Hessen konkrete Zahlen zur Landschaftszerschneidung. Sie sind eine wichtige Grundlage für vergleichende Untersuchungen in anderen Bundesländern. Dem von Jaeger et al. (2001) langfristig geforderten Ziel, bundesweite Vergleichsdaten zu schaffen, ist man somit ein Stück näher gekommen.

Eine historische Trendanalyse geht über eine Beschreibung des aktuellen Zustands hinaus. Sie ist methodisch bedingt allerdings sehr zeitaufwendig, insbesondere durch den Arbeitsaufwand bei der Digitalisierung historischer Karten. Dies mag ein Grund dafür sein, dass Trendanalysen bisher nur in wenigen Bundesländern durchgeführt wurden (vgl. Esswein et al 2002, Voerkel 2005). Entwicklungen im Zeitverlauf zu dokumentieren, ist jedoch die wesentliche Grundlage für ein landesweites Monitoring und eine wichtige Ausgangslage für zukünftige Umweltbeobachtungen. Mit einer historischen Dokumentation können Zukunftsszenarien entworfen werden, wie sich das Verkehrswegenetz bei unveränderter Politiklage weiter entwickeln kann. Da sich die zunehmende Zerschneidung schleichend vollzieht (Renn et al. 2000) und die Folgen eines Straßenneubaus meist nur lokal bewertet und nicht in ihrem landschaftsökologischen Kontext gesehen werden, spielen solche Negativszenarien eine nicht zu unterschätzende Rolle bei der Entscheidungsunterstützung in Verkehrs- und Landschaftsplanung.

Durch eine Trendanalyse ist es nicht nur möglich, den Zustand verschiedener Teilräume miteinander zu vergleichen. Es können vielmehr Räume identifiziert werden, in denen die Entwicklung in der Vergangenheit besonders dramatisch verlief. An Ökosysteme mit konstant erschwerten Bedingungen, wie z.B. in der Nähe von Siedlungen, können sich einige Tierarten im Laufe der Zeit anpassen. Starke Veränderungen in relativ kurzen Zeiträumen bedeuten dagegen einen hohen Stresspegel, der schwerer zu kompensieren ist, und eine hohe Anpassungsfähigkeit erfordert. Regionen, die in der

Vergangenheit besonders störungsintensiv waren, sind gegenüber weiteren Veränderungen äußerst empfindlich, da die Anpassung der lokalen Flora und Fauna an Störungen unter Umständen noch nicht abgeschlossen ist. Es ist wichtig, solche Regionen zu identifizieren und sie von weiteren Verkehrsplanungen in naher Zukunft möglichst auszunehmen.

Als positive Entwicklung in diese Richtung ist daher das Ergebnis der Länderinitiative für einen gemeinsamen Satz von Kernindikatoren (LIKI) zu werten. Innerhalb dieses Rahmens trafen sich im Jahr 2004 Vertreter aus 14 Bundesländern um gemeinsam mit dem Umweltbundesamt (UBA), dem Bundesamt für Naturschutz (BfN) sowie Vertreter(inne)n aus der Wissenschaft (ETH Zürich und Universität Stuttgart) an einer Vereinheitlichung des Indikators Landschaftszerschneidung zu arbeiten. Mit den erzielten Ergebnissen wird es in Zukunft möglich sein die Entwicklung der Landschaftszerschneidung nach einheitlichen Kriterien zu erfassen, um somit für alle Bundesländer vergleichbare Zeitreihen für den Indikator Landschaftszerschneidung darzustellen (vgl. auch Schupp 2005).

Mit landesweiten Trendanalysen zur Landschaftszerschneidung wird es auch möglich sein, bisher nicht beantwortete Fragen zu den Effekten von Strassen auf Tierpopulationen zu beantworten. Es ist relativ viel darüber bekannt, wie einzelne Tierarten lokal auf Straßen reagieren (Glitzner et al. 1999), doch die Reaktion von Populationen und Metapopulationen auf eine landesweite Verdichtung des Verkehrswegenetzes ist bislang wenig beforscht. Findlay und Bourdages (2000) konnten nachweisen, dass Tierarten mit erheblichen Zeitverzögerungen auf den Neubau von Straßen reagieren. Wenn landschaftsökologische Untersuchungen nur vom aktuellen Zustand der Zerschneidung ausgehen, können Aussagen über die ökologischen Effekte erst in weiter Zukunft getroffen werden. Um jedoch baldmöglichst zu statistisch abgesicherten Ergebnissen kommen zu können, ist der Blick in die Vergangenheit unverzichtbar.

Ein Projekt zu den landschaftsökologischen Effekten der Zerschneidung auf Tierpopulationen läuft derzeit am Institut für Biometrie und Populationsgenetik der Universität Giessen. Hier werden die für Hessen nun vorliegenden, historischen Zerschneidungskarten mit aktuellen und soweit vorhanden auch mit historischen Bestandsdaten von Tierarten überlagert. Ziel ist es, den Einfluss der Zerschneidung auf die landesweiten Bestandsrückgänge von Tierarten zu quantifizieren. Die Flächenzerschneidung wird dabei im landschaftsökologischen Kontext gesehen. Als Ergebnis soll die Einflussschwere der Landschaftszerschneidung im Verhältnis zu anderen Standortparametern (Witterung, Bodenverhältnisse, historische Landnutzungsänderungen) gewichtet werden. Wenn Korrelationen zwischen den Werten des Zerschneidungsindices und der Vorkommensdichte von Zielarten nachgewiesen werden können, erhält die Effektive Maschenweite eine verstärkte Indikatorqualität bei der ökologischen Bewertung von Landschaften.

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Chapter 4.
Effekte der Landschaftszerschneidung auf die Unfallhäufigkeit und Bestandsdichte von Wildtierpopulationen

Zur Indikationsqualität der Effektiven Maschenweite

Roedenbeck IA, und Köhler W. 2006. Effekte der Landschaftszerschneidung auf die Unfallhäufigkeit und Bestandsdichte von Wildtierpopulationen. Zur Indikationsqualität der Effektiven Maschenweite. *Naturschutz und Landschaftsplanung* 38(10/11): 314-32.

Zusammenfassung

In jüngster Vergangenheit hat eine rasante Entwicklung von Indices zur Messung der Landschaftszerschneidung stattgefunden -besondere Bekanntheit erlangte in Deutschland die Effektive Maschenweite (m_{eff}). Indikatoren werden zur Bewertung von Veränderungen der Landschaft benötigt und im Umweltmonitoring eingesetzt. Der Kern jeder Bewertung ist die Qualität der Indikation. Die vorliegende Untersuchung prüft und diskutiert die Indikationsqualität von m_{eff}. Es wird die Frage gestellt, ob der Index darauf beschränkt ist, quantitativ den Grad der Zerschneidung zu messen, oder auch in der Lage ist, einen qualitativen Zusammenhang zu einem ökologischen Prozess in der Landschaft anzuzeigen - z.B. zur Überlebensfähigkeit von Wildtierpopulationen in zerschnittenen Lebensräumen. (1) In einer landesweiten, quantitativen Bestandsanalyse wird die zunehmende Verdichtung des Verkehrs- und Siedlungsnetzes in Hessen von 1930 bis 2002 dokumentiert. (2) Trends der Bestandsentwicklung und Unfallhäufigkeit von Reh, Wildschwein, Fuchs und Dachs werden von 1959 bis 2003 anhand der hessischen Streckenlisten aufgezeigt. (3) Der aktuelle Zerschneidungsgrad der hessischen Landkreise wird mit den Streckenlisten verglichen: Je stärker der Zerschneidungsgrad eines Landkreises, desto geringer der aktuelle Bestand und desto höher die Unfallrate. (4) Die Bedeutung der Landschaftszerschneidung wird im Verhältnis zu anderen Habitatfaktoren beispielhaft für Rehe beleuchtet. Im Anschluss wird diskutiert, ob und in welchem Ausmaß die hier gefundenen Ergebnisse die Indikationsfähigkeit der Effektiven Maschenweite inhaltlich erweitern. Die vorgestellten Ergebnisse liefern einen ersten Hinweis, dass Straßen auf Landschaftsebene Wildtierbestände beeinflussen. Es wird jedoch dringend dazu geraten, m_{eff} nicht als alleiniges Kriterium, sondern neben qualitativen Indikatoren, z.B. zur artspezifischen Habitatqualität und -verfügbarkeit, im Umweltmonitoring einzusetzen.

Abstract

Effects of Landscape Fragmentation on Road Mortality Rates and Abundance of Wildlife Populations - Indication quality of effective mesh size. *The last years have seen a rapid development of new indices for measuring landscape fragmentation – with the indicator 'effective mesh size' (m_{eff}) reaching particular importance. There is a large need of such indicators to evaluate landscape changes and for environmental monitoring. Core of each evaluation is the quality of the indication. The study presented investigates and discusses the indication quality of m_{eff}. We analyse if the index is limited to quantify the degree of fragmentation, or if it is capable to identify a qualitative correlation to an ecological process in the landscape, e.g. the survival probability of wildlife populations in fragmented landscapes. (1) A state-wide, quantitative inventory documents the increasing densification of the traffic and settlement network in the Federal State of Hesse between 1930 and 2002. (2) We show trends of population development and road mortality frequencies of roe deer, wild boar, fox and badger between 1959 and 2003 using the Hessian harvest reports. (3) We compare the pesent degree of fragmentation of the Hessian districts (m_{eff}) with the harvest reports: the higher the degree of fragmentation, the lower abundance and the higher road mortality rates. (4) The relative importance of landscape fragmentation versus other habitat factors is exemplified for roe deer. Finally, we discuss whether or not, and to what extent the results represent an extension of the indication ability of 'effective mesh size'. The results presented provide first evidence that roads affect wildlife populations at the landscape scale. We recommend not to use m_{eff} as exclusive criterion but to combine it with other qualitative criteria in environmental monitoring programs, such as species-specific habitat quality and availability.*

4.1 EINLEITUNG

Straßen beeinträchtigen wildlebende Tierarten (Forman et al. 2003, Glitzner et al. 1999, Jackson 2000, Sherwood et al. 2002, Spellerberg 2002, Trombulak und Frissell 2000, Underhill und Angold 2000). Sie wirken dabei auf verschiedenen Raumebenen und betreffen unterschiedliche Ausschnitte (bzw. Ebenen) einer Population: Die Mortalität spielt sich direkt auf der Straße ab, wenn Tiere beim Überquerungsversuch vom Verkehr erfasst werden, auf der Straße verenden oder sich verletzt in straßennahe Areale schleppen und dort verenden. Vom tödlichen Zusammentreffen mit Fahrzeugen sind zunächst Individuen betroffen. Die populationswirksamen Effekte der Verkehrsmortalität sind noch unklar (Eichstädt und Roth 1997). Der direkte Habitatverlust kommt im Zuge der Bauphase zum Tragen und betrifft die Habitatfläche, die durch Baufahrzeuge beeinträchtigt ist oder später asphaltiert und damit versiegelt wird. Der indirekte Habitatverlust betrifft angrenzende Gebiete, die durch (Lärm-, Staub-, Schwermetall-, Licht-) Immissionen in ihrer Habitatqualität beeinträchtigt werden. Die Effekte solcher Störungen sind geringere Brutdichten und damit ein verminderter Reproduktionserfolg der lokalen Population. Die Zerschneidung von Lebensräumen und die damit einhergehende Isolation können zur Unterschreitung von Mindestarealen führen, die eine (Meta-) Population zum Überleben braucht. Isolation führt zu genetischen Drift und damit zur verminderten Anpassungsfähigkeit einer Population an Umweltschwankungen. Das Aussterberisiko der Population wird damit erhöht. Straßen wirken also auf unterschiedlichen Raumebenen. Je größer die Effektreichweite, desto größer der Populationsausschnitt, der betroffen ist.

Es schließt sich die Frage an, welche Effektreichweiten bisher als wissenschaftlich abgesichert gelten. In der Literatur finden sich – neben den zahlreichen Review-Artikeln zu den Effekten von Straßen auf Tierarten (Glitzner et al. 1999, Jackson 2000, Trombulak und Frissell 2000, Underhill und Angold 2000) – zunächst eine Reihe von Untersuchungen, die sich mit der Verkehrsmortalität befassen (z.B. Lodé 2000, Hauer et al. 2002, Reeve und Huijser 1999). Es folgen Studien zum indirekten Habitatverlust. Hier werden oft einzelne Straßen betrachtet und Nachweise erbracht, dass diese durch verschiedene Ursachen (insb. Lärm) den Reproduktionserfolg einer Population beeinträchtigen (Illner 1992, Huijser und Bergers 2000, Van der Zande 1980, Fahrig 1995, Reijnen et al. 1995 und 1996). Studien zum Effekt von Straßen auf Landschaftsebene sind selten. Die vorhandenen landschaftsökologischen Untersuchungen beziehen sich fast ausschließlich auf Amphibienpopulationen (Carr und Fahrig 2001, Findlay und Bourdages 1999, Findlay und Houlahan 1997, Löfvenhaft 2004, Vos und Chardon 1998). Für größere Säuger wurde nachgewiesen, dass Tiere langjährig etablierte Raumnutzungsmuster ändern, um Straßen auszuweichen (Rost und Bailey 1979, Witmer und DeCalesta 1985, Thurber et al. 1994). Ob diese Verhaltensänderungen auch Effekte auf die Überlebensfähigkeit von Populationen haben, konnte bislang nicht nachgewiesen werden.

Das Ungleichgewicht zwischen lokalen und landschaftsökologischen Untersuchungen hat methoden-historische Gründe. Zur Interpretation von ökologischen Zusammenhängen, die sich auf Landschaftsebene abspielen, müssen analysestarke Handwerkszeuge zur Verfügung stehen, die großflächige, raumbezogene Informationen verarbeiten können. Da entsprechende Werkzeuge in der Vergangenheit fehlten und landschaftsbezogene Informationen nur mit enormem Aufwand zu erheben waren, beschränkten sich ältere Studien meist auf die lokale Ebene einzelner Straßen. Mit der rasanten Entwicklung von Geo-Informations-Systemen (GIS) stehen inzwischen Werkzeuge zur Verfügung, die die Verarbeitung und räumliche Abfrage von Landschaftsinformationen ermöglichen.

Parallel zur Entwicklung von GIS ist eine sprunghafte Entwicklung von Landschaftsindices vonstatten gegangen (Rutledge 2003, Rutledge und Miller 2005), die z.B. im GIS-Tool FRAGSTATS implementiert sind (McGarigal et al. 1995). Landschaftsindices beschreiben die Struktur einer Landschaft, z.B. die räumliche Verteilung von Habitatpatches und deren Fragmentierungsgrad, die räumliche Verteilung von Straßen, oder die Dichte des Verkehrswegenetzes. Beispiele für Zerschneidungsindices sind die Anzahl unzerschnittener, verkehrsarmer Räume UZR (BfN 1999) und die Effektive Maschenweite m_{eff} (Jaeger 2000). Solche Zerschneidungsindices finden in jüngster Vergangenheit nicht selten Verwendung in indikatorbasierten Umweltbewertungen (Schupp 2005). Umweltbewertungssysteme ermöglichen es, Landschaften zu beobachten und bedrohliche Veränderungen rechtzeitig wahrzunehmen, bzw. über die Einführung von Schwellenwerten negative Trendentwicklungen zu verhindern. Der Kern jeder indikatorbasierten Umweltbewertung ist die Qualität der Indikation. Ein Zerschneidungsindex sollte Entwicklungen der Landschaftszerschneidung anzeigen können. Seine Indikationsqualität wäre darüber hinaus deutlich erhöht, wenn er eine konkrete Umweltqualität bzw. Landschaftsfunktion indizieren würde, wie zum Beispiel ein vielfältiges Landschaftsbild, oder eine artenreiche Flora und Fauna (Biodiversität). Die Interpretation gängiger Zerschneidungsindices als qualitativer Indikator wird meist wie folgt begründet: Es ist bekannt, dass Straßen lokal negative Effekte auf Arten haben. Folglich ist anzunehmen, dass ein Zerschneidungsindex - der die Anordnung vieler Straßen in der Landschaft beschreibt - den Bestand einer Art anzeigt. Es findet also eine Extrapolation vom Punkt auf die Fläche und vom Individuum auf die Population statt, die unzulässig ist. Um diese Extrapolation zu rechtfertigen, müsste der Nachweis erbracht werden, dass Straßen auf Landschaftsebene tatsächlich Effekte auf Populationen haben. Anders gesagt: Um die Indikationsfähigkeit eines Zerschneidungsindices um eine qualitative Komponente (z.B. die Überlebenswahrscheinlichkeit von Tierpopulationen als Grundvoraussetzung für die Artenvielfalt) zu erweitern, muss der Nachweis erbracht werden, dass zwischen den Werten des Indices und dem Bestand von Tierarten tatsächlich Zusammenhänge bestehen, und zwar auf der gesamten betrachteten Raumskala.

Mit der hier vorgestellten Untersuchung soll geprüft werden, ob der Zerschneidungsindex Effektive Maschenweite (m_{eff}) neben dem Zerschneidungsgrad der Landschaft auch die Unfallhäufigkeit bzw. Bestandssituation von Wildtierarten indizieren kann. Es werden vier Arbeitsziele verfolgt:

(1) Die Entwicklung der Landschaftszerschneidung in Hessen wird in sechs Zeitschritten von 1930-2003 dokumentiert. (2) Trends in Bestandsentwicklung und Unfallhäufigkeit von vier hessischen Wildtierarten (Reh, Wildschwein, Fuchs und Dachs) werden von 1959 bis 2003 aufgezeigt. (3) Zur Aufklärung räumlicher Muster wird der aktuelle Zerschneidungsgrad auf Ebene der hessischen Landkreise mit Bestandsdichten und Unfallhäufigkeiten der vier Wildtierarten verglichen. (4) Am Beispiel von Rehen wird die Bedeutung der Landschaftszerschneidung im Verhältnis zu anderen Habitatfaktoren untersucht.

Im Anschluss wird diskutiert, welche Aussagekraft die hier präsentierten Ergebnisse besitzen. Die zu beantwortende Kernfrage ist: Wenn Wertänderungen eines Zerschneidungsindices mit Bestandsänderungen einer Zielart einhergehen, wäre damit die Indikationsfähigkeit des Indices um eine qualitative Komponente erweitert?

4.2 Material und Methoden

4.2.1 Der Zerschneidungsindex

Als Zerschneidungsindex dient in dieser Studie die effektive Maschenweite m_{eff} (Jaeger 2000). Der Index ist ein Ausdruck für die Möglichkeit, dass sich zwei Tiere, die zufällig und unabhängig voneinander in einem Gebiet ausgesetzt werden, begegnen können, wobei die Begegnungswahrscheinlichkeit mit einer zunehmenden Anzahl von Barrieren in der Landschaft sinkt. Eine Barrierewirkungen für die Raumbewegung der hier untersuchten Arten wurde folgenden Landschaftsstrukturen zugesprochen: Straßen (Kategorien Kreisstraße bis Autobahn), Bahnlinien, Siedlungen, Seen und Flüssen über 6m Breite. Datengrundlage für die aktuelle Landschaftssituation war das ATKIS-Basis DLM 2 (Stand 2002). Die historische Entwicklung der Bahnlinien, Siedlungen und Straßen wurde aus historischen topographischen Übersichtskarten (bzw. Karten des Deutschen Reiches 1930) entnommen.

Für die Berechnung der effektiven Maschenweite werden zwei Angaben benötigt: (1) Die Gesamt-Flächengröße F_g eines untersuchten Gebietes und (2) die Flächengröße F_i jedes i-ten unzerschnittenen Teilraums in dem untersuchten Gebiet.

$$m_{eff} = F_g * \sum_{i=1}^{n} \left(\frac{F_i}{F_g} \right)^2 = \frac{\sum F_i^2}{F_g}$$

Der Index m_{eff} wurde für alle hessischen Landkreise (N=26) zu allen sechs Zeitschritten berechnet (1930, 1966, 1977, 1989, 1995 und 2002). Außerdem wurde für jeden

Landkreis der Wert m_{eff} [Trend$_{1930-2002}$] berechnet, der die Intensität der Infrastrukturentwicklung widerspiegelt. Er berechnet sich als prozentualer Anteil der Maschenweite 2002 vom Basisjahr 1930:

$$m_{eff}[Trend_{1930-2002}] = 100 - \frac{m_{eff}[2002]*100}{m_{eff}[1930]}$$

Sämtliche raumbezogenen Daten wurden mit der GIS-Software ArcView 3.2 und ArcGIS 9.0 ausgewertet.

4.2.2 LANDSCHAFTSZERSCHNEIDUNG UND WILDBESTAND

Zur Dokumentation von Trends in der Bestandsentwicklung und Unfallhäufigkeit von Reh (*Capreolus capreolus*), Wildschwein (*Sus scrofa*), Fuchs (*Vulpes vulpes*) und Dachs (*Meles meles*) wurden die hessischen Streckenlisten auf Landesebene verwendet, die jeder Jagdrevierinhaber auf Revierebene führt und jährlich an die Jagdbehörden meldet (die Zählperiode geht von April bis März des Folgejahres). In den Streckenlisten wird pro Jahr die Anzahl der geschossenen Individuen einer Art dokumentiert, wobei sich der Abschuss, dem Prinzip der Nachhaltigkeit zufolge, am Bestand orientieren sollte. Außerdem enthalten sie als so genannte „Fallwildzahlen" die Anzahl der im Verkehr tot aufgefundenen bzw. gemeldeten Individuen. Die Jagdstrecke war für alle vier Arten von 1959 bis 2003 landesweit verfügbar, die Fallwildangaben standen nur für Reh und Wildschwein seit 1959 zur Verfügung.

Zur Analyse räumlicher Muster der aktuellen Bestandssituation und Unfallhäufigkeit wurden die Streckenlisten der Jahre 2001/02 (und 1995/96 als Kontrolle) auf Landkreisebene verwendet, da beide Jahrgänge annährend vollständig für alle hessischen Landkreise vorliegen. Es wurden zwei Parameter berechnet: (1) Die Anzahl der geschossenen Individuen je Landkreis, bezogen auf die Landkreisfläche. Dieser Parameter wird im weiteren Verlauf als Bestand bezeichnet. (2) Der flächengewichtete Anteil Fallwild an der Jagdstrecke. Dieser Parameter wird im weiteren Verlauf als Unfallrate bezeichnet. Es wurde bewusst nicht die reine Fallwildanzahl gewählt, um den Faktor „je mehr Tiere im Landkreis, desto mehr Fallwild" ausklammern zu können.

Zwischen den aktuellen Beständen, bzw. Unfallraten, und den m_{eff}- Werten aller sechs Zieljahre wurden lineare Regressionen durchgeführt (2 Jagdjahre x 6 m_{eff}-Jahre x 2 Bestandsindices = 24 Regressionen je Art). Jeder Regression liegt eine Stichprobe von 24-26 Landkreisen zugrunde. Sofern ein nicht-linearer Zusammenhang bestand, wurden die Werte vor der Regressionsanalyse durch doppel-log-Transformation linearisiert (Köhler et al. 2002).

4.2.3 LANDSCHAFTSZERSCHNEIDUNG UND LANDNUTZUNG

Am Beispiel von Rehen wurde untersucht, wie Landschaftszerschneidung und die Flächenanteile verschiedener Landnutzungsformen den aktuellen Bestand gemeinschaftlich beeinflussen. Hierfür wurden Daten vom Statistischen Landsamt (Wiesbaden) auf Landkreisebene zur Verfügung gestellt: Aus der Bodennutzungshaupterhebung des Jahres 2000 die Siedlungsfläche (Gebäude und Gewerbefläche incl. Freiflächen), Waldfläche und landwirtschaftlich genutzten Fläche; aus der Agrarstrukturerhebung von 2002 die Flächenanteile von Ackerland, Brache und Dauergrünland. Aus dem ATKIS-Basis-DLM 25 (Stand 2002) wurde die Dichte von Überlandstraßen (Kreisstraße bis Autobahn) je Landkreis [km/km²] berechnet.

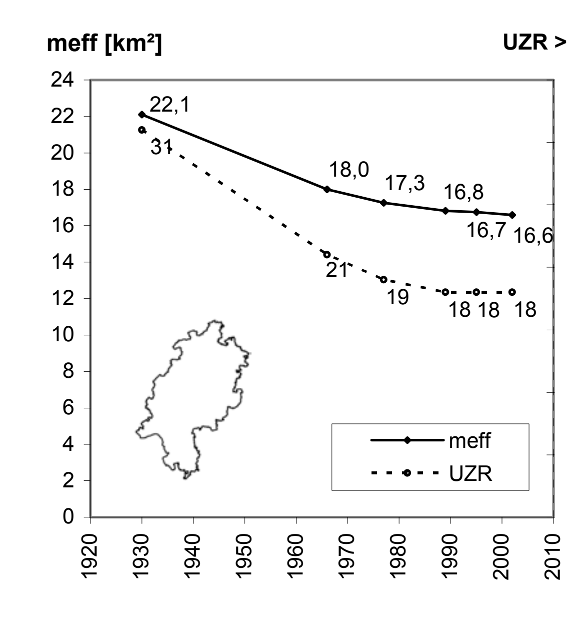
Um die Einflussvariablen in Gruppen zu klassifizieren und Zusammenhänge zwischen mehreren Einflussgrößen darzustellen wurden eine Hauptkomponentenanalyse, eine Clusteranalyse und schrittweise multiple Regressionen durchgeführt. Zur detaillierten Darstellung der statistischen Methoden wird auf Köhler et al. (2002) und Brosius (1998) verwiesen. Sämtliche statistischen Analysen erfolgten mit Hilfe des Softwarepakets SPPS (Version 11.5.1).

4.3 ERGEBNISSE

4.3.1 TREND DER LANDSCHAFTSZERSCHNEIDUNG IN HESSEN

Die Landschaftszerschneidung durch Verkehrsinfrastrukturen und Siedlungen hat in Hessen seit 1930 kontinuierlich zugenommen (**Fig. 4.1**). Von 1930 bis 2002 sinkt der Wert der effektiven Maschenweite von 22,10 auf 16,59 km². Dies entspricht einer Zunahme der Landschaftszerschneidung um rund 25% in knapp 70 Jahren. Die Anzahl der Räume größer 50 km² sinkt von 1930 bis 2002 von 31 auf 18 - ein Rückgang von rund elf auf rund sechs Prozent der Landesfläche. Die Landschaftsfragmente im Netz werden kleiner und nehmen zahlenmäßig von 6.658 auf 10.458 zu.

Der Mittelwert der effektiven Maschenweite aller Landkreise beträgt 11,81 km² im Jahr 2002 und 15,86 km² im Jahr 1930. In allen Landkreisen hat die Landschaftszerschneidung seit 1930 zugenommen, allerdings unterschiedlich stark. Die stärksten infrastrukturellen Entwicklungen haben sich im Rhein-Main-Gebiet und auf den ackerbaulich geeigneten Böden der hessischen Flusstäler abgespielt, weniger betroffen waren die peripheren Mittelgebirgslagen (für Details s. Roedenbeck et al. 2005). **Fig. 4.2** belegt den Zusammenhang zwischen m_{eff} [Trend$_{1930\text{-}2002}$] und den m_{eff}- Werten von 2002. Landkreise mit einer starken infrastrukturellen Entwicklung in den vergangenen 70 Jahren zeigen aktuell die niedrigsten m_{eff}- Werte und damit den höchsten Zerschneidungsgrad. Aufgrund des Zusammenhangs beider Parameter werden die Wildbestandsdaten nur mit den m_{eff}- Werten und nicht mit den m_{eff} [Trend]- Werten verglichen.



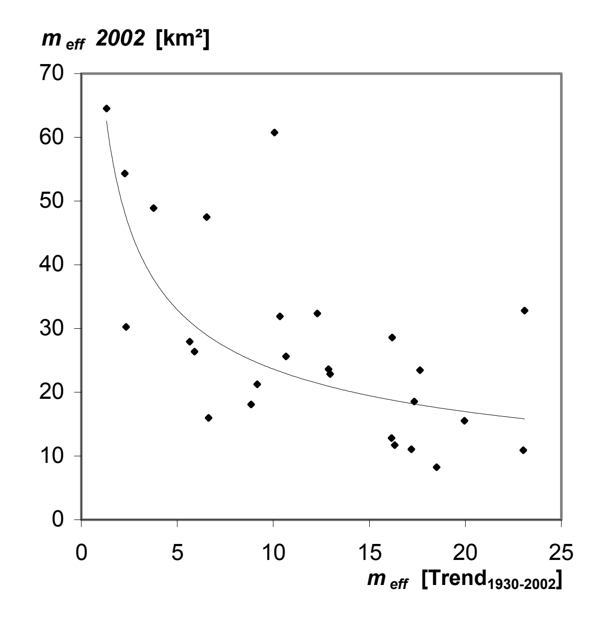


***Fig. 4.1 (links)**: Entwicklung der Landschaftszerschneidung in Hessen von 1930 bis 2002. Gezeigt wird die effektive Maschenweite (m_{eff}) und die Anzahl der unzerschnittenen Räume größer 50 km² (UZR>50).*

***Fig. 4.2 (rechts)**: Zusammenhang zwischen dem aktuellen Zerschneidungsgrad der 26 hessischen Landkreise (m_{eff} 2002) und der Stärke der infrastrukturellen Entwicklung von 1930 bis 2002 (m_{eff} [Trend$_{1930\text{-}2002}$]). Die Korrelation beider Parameter ist signifikant mit $p<0{,}01$ und $r= 0{,}624$.*

4.3.2 Trends in Bestandsdichten und Unfallhäufigkeiten von Wildtieren

Die Jagdstrecken von Reh, Wildschwein, Fuchs und Dachs steigen in den letzten Jahren mehr oder weniger konsequent an (**Fig. 4.3**). Die Rehstrecke erreichte um 1969 ein kurzzeitiges Tief, steigt seit dem aber kontinuierlich und stagniert seit ca. 1991 auf konstant hohem Niveau von über 60.000 Individuen pro Jagdjahr. Die Wildschweinstrecke steigt nach Abschusstiefen in den 60er Jahren seit 1982 mehr oder weniger stark an und hat im Jahr 2001 einen absoluten Höhepunkt erreicht. Die Fuchsstrecke übersteigt 1990 die Grenze von 30.000 Individuen und erreicht 1995 einen vorläufigen Höhepunkt. Die Dachsstrecke hat sich nach einem von 1967 bis 1981 andauernden Tiefpunkt wieder erholt und steigt seit 1982 auf im Jahr 2003 knapp 3000 Individuen.

Mit zunehmenden Beständen steigt über die Jahre die Anzahl der im Verkehr getöteten Individuen. Die Entwicklung der Bestände und Fallwildzahlen über die Zeit korreliert bei allen vier Arten hoch signifikant positiv (Reh: $r=0{,}77$; $p<0{,}001$; $N=45$/ Wildschwein: $r=0{,}97$; $p<0{,}001$; $N=45$/ Dachs: $r=0{,}96$; $p<0{,}001$; $N=18$/ Fuchs: $r=0{,}93$; $p<0{,}001$; $N=17$).

Wenn die Unfallhäufigkeit allerdings allein von den Bestandsdichten abhängig wäre, müsste die Unfallrate (hier angegeben als Quotient aus Fallwild und Jagdstrecke) über die Jahre hinweg konstant bleiben. Die Unfallrate bei Rehen und Wildschweinen nimmt aber seit 1959 mit mehr oder weniger starken Schwankungen kontinuierlich zu, was bestandsunabhängige Einflussgrößen vermuten lässt. Andere für den Anstieg der

Unfallzahlen verantwortliche Parameter mögen die zunehmende Verkehrsnetzdichte und die linear steigenden PKW-Dichten in Hessen sein (**Fig. 4.1**). Die zeitlichen Schwankungen in den Unfallraten sind ebenfalls nicht durch Schwankungen in den Streckenzahlen zu erklären (vgl. **Fig. 4.3** und **Fig. 4.4**), sondern müssen andere, z.B. verkehrsbedingte Ursachen haben.

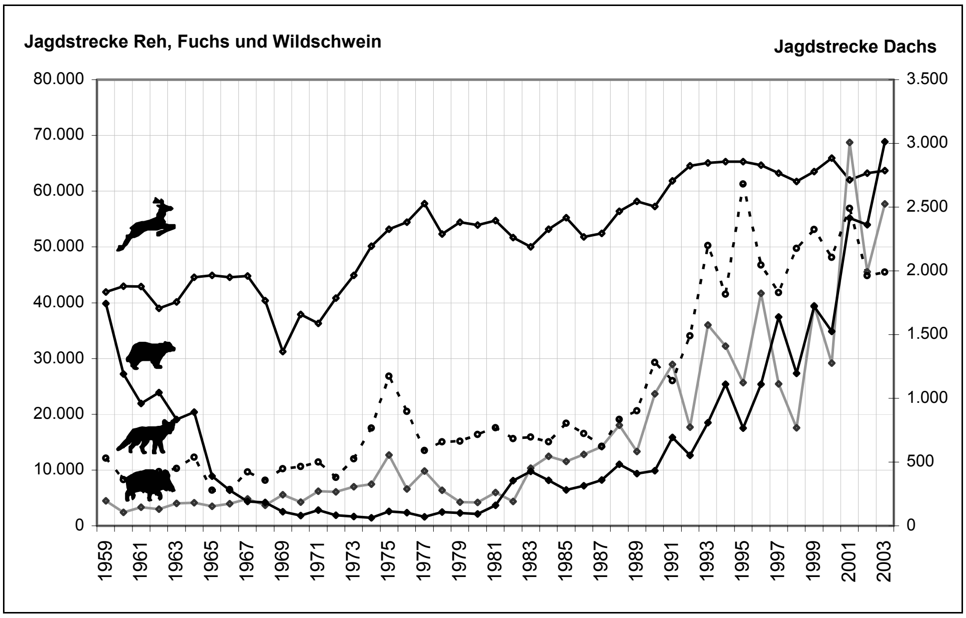


Fig. 4.3: *Hessische Jagdstrecke für Reh, Wildschwein, Fuchs und Dachs von 1959-2003. Abgebildet ist die Zahl der geschossenen Individuen je Art im Zeitverlauf – ohne Fallwild. Quelle: Oberste Jagdbehörde Hessen (HMULV).*

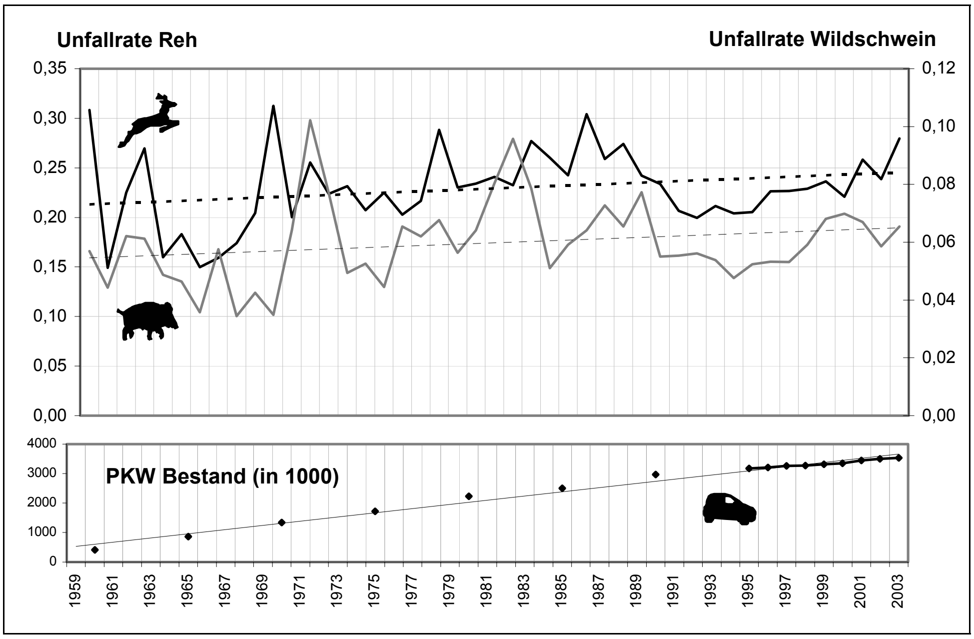


Fig. 4.4*: Entwicklung des PKW-Bestands in Hessen sowie der Unfallraten (Anteil Fallwild an der Jagdstrecke) von Rehen und Wildschweinen 1959-2003 (Datengrundlage für Berechnung: Oberste Jagdbehörde Hessen - HMULV).*

4.3.3 Räumliche Muster in Bestandsdichten und Unfallraten

Zur Analyse räumlicher Muster von Bestandsdichten und Unfallraten wurde der aktuelle Bestand und aktuelle Unfallraten der vier Wildtierarten (Jagdjahr 2001/02) mit dem aktuellen Zerschneidungsgrad der hessischen Landkreise (Stand 2002) verglichen. Lineare Regressionsanalysen zeigen (Fig. 4.5, Tab. 4.1):

(1) Je stärker der Zerschneidungsgrad eines Landkreises, desto höher die Unfallrate von Reh, Wildschwein und Fuchs (Reh: $r^2=0{,}53$; $p<0{,}001$; N=24/ Wildschwein: $r^2=0{,}81$; $p<0{,}001$; N=23/ Fuchs: $r^2=0{,}38$; $p<0{,}01$; N=24). Der Zusammenhang ist bei Rehen linear, die Werte von Wildschwein und Fuchs wurden vor der Regressionsanalyse log-transformiert. Für Dachse ergibt sich kein signifikanter Zusammenhang.

(2) Je stärker der Zerschneidungsgrad eines Landkreises, desto geringer die aktuellen Bestände von Reh, Schwarzwild, Fuchs und Dachs (Reh: $r^2=0{,}65$; $p<0{,}001$; N=24/ Wildschwein: $r^2=0{,}31$; $p<0{,}01$; N=24/ Fuchs: $r^2=0{,}49$; $p<0{,}001$; N=24/ Dachs: $r^2=0{,}49$; $p<0{,}001$; N=23). Der Zusammenhang ist für Reh und Wildschwein linear, die Werte von Fuchs und Dachs wurden vor der Regressionsanalyse log-transformiert.

Die Bestandszahlen von Fuchs und Dachs sowie der Unfallraten von Fuchs und Wildschwein zeigen, dass ein nicht-linearer Zusammenhang zur Landschaftszerschneidung besteht. Die Parameter der Regressionsfunktionen wurden daher mit Hilfe der doppel-log-Transformation angepasst (Köhler et al. 2002). Diese Arten zeigen in Räumen mit geringer Zerschneidung wenig Reaktionen, während bei starker Zerschneidung die Unfallraten sprunghaft ansteigen, bzw. der Bestand sprunghaft abnimmt. Dieser Reaktionstypus weist auf einen *Schwellenwert* der Landschaftszerschneidung hin, bei dem die untersuchten Wildtierarten verkehrsbedingte Verluste bzw. Isolationseffekte möglicherweise schwerer kompensieren können. Obwohl die kleine Stichprobengröße nur eine Annährung an den Verlauf der Trendlinie zulässt, ist bei Wildschwein und Fuchs eine Abschätzung zur Lage des Schwellenwerts möglich. Er liegt zwischen m_{eff}- Werten von 8 bis 10 km² (**Fig. 4.5**). Ein Vergleich der aktuellen Bestandsdichten mit der historischen Zerschneidungssituation der Landkreise zeigt, dass der aktuelle Bestand mit allen historischen Jahren korreliert ist (**Tab. 4.1**). Dieses Ergebnis ist zu erwarten, da die Landschaftszerschneidung langsam zunimmt und die Verhältnismäßigkeiten der Landkreise untereinander annährend gleich bleiben. Interessant ist allerdings, dass der aktuelle Bestand (2001/02) nicht immer mit der aktuellen Landschaftszerschneidung am Engsten zusammenhängt. Bei Rehen, Fuchs und Dachs korreliert der Bestand am Engsten mit der Zerschneidung im Jahr 1968.

4.3.4 Landschaftszerschneidung und Landnutzung auf Landkreisebene

Die Bedeutung der Landschaftszerschneidung im Verhältnis zu anderen Habitatfaktoren soll beispielhaft für Rehe beleuchtet werden. Einfache lineare Regressionen zwischen dem Rehbestand und den Flächenanteilen der Landnutzungstypen zeigen, dass den stärksten negativen Einfluss auf den Rehbestand die Siedlungsfläche hat ($p<0{,}001$). Mit abnehmender Stärke folgen Straßendichte ($p<0{,}001$) und der Grad der Landschaftszerschneidung ($p<0{,}001$). Einen positiven Einfluss auf den Rehbestand haben Waldfläche ($p<0{,}001$), Dauergrünland ($p<0{,}01$) und landwirtschaftlich genutzte Fläche ($p<0{,}01$). Acker und Brache stehen in keinem signifikanten Zusammenhang zum Rehbestand (**Tab. 4.2**).

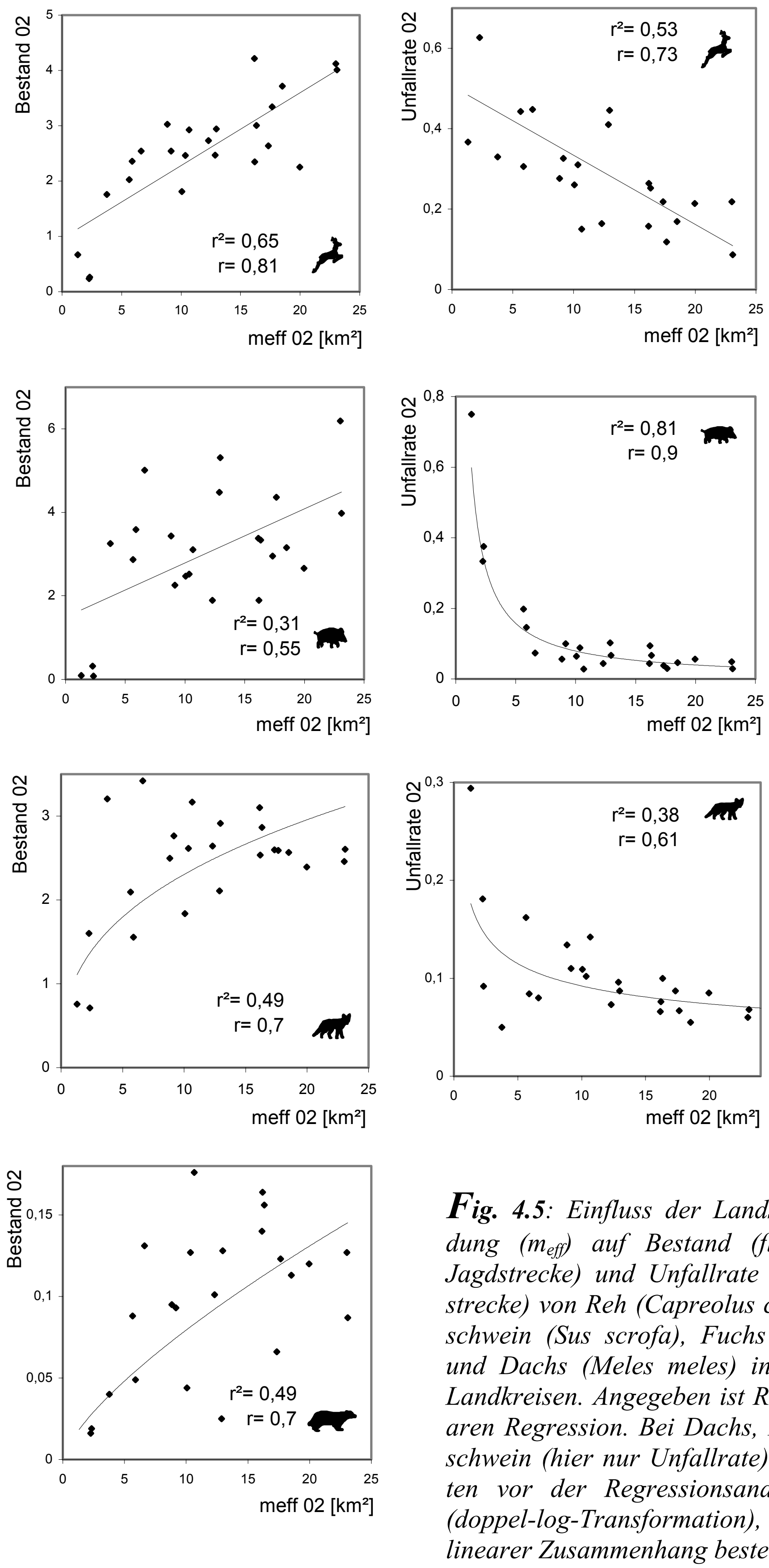


***Fig. 4.5**: Einfluss der Landschaftszerschneidung (m_{eff}) auf Bestand (flächengewichtete Jagdstrecke) und Unfallrate (Fallwild/ Jagdstrecke) von Reh (Capreolus capreolus), Wildschwein (Sus scrofa), Fuchs (Vulpes vulpes) und Dachs (Meles meles) in den hessischen Landkreisen. Angegeben ist R und R² der linearen Regression. Bei Dachs, Fuchs und Wildschwein (hier nur Unfallrate) wurden die Daten vor der Regressionsanalyse linearisiert (doppel-log-Transformation), weil ein nicht-linearer Zusammenhang besteht.*

Tab. 4.1: *Ergebnisse der Regressionsanalyse mit m_{eff} je Landkreis (1930-2002) als unabhängige Variable und Bestand (Jagdstrecke pro ha) bzw. Unfallrate (Fallwild/ Jagdstrecke) als Zielvariablen. Angegeben sind die Vorzeichen der Regressionskoeffizienten,R als Maß für die Güte der Anpassung und die Signifikanzniveaus (***: p<0,001/ **: p<0,01/ *: p<0,05/ ns= nicht signifikant). [1)]: Variablen wegen Nicht-Linearität ln-transformiert.*

Art			**Bestand**				**Unfallrate**		
			2001/02		**1995/96**		**2001/02**		**1995/96**
Reh	m_{eff} **2002**	+	0,805 ***	+	0,624 ***	-	0,725 ***	-	0,757 ***
	m_{eff} **1995**	+	0,804 ***	+	0,623 ***	-	0,725 ***	-	0,754 ***
Capreolus capreolus	m_{eff} **1988**	+	0,803 ***	+	0,622 ***	-	0,726 ***	-	0,753 ***
	m_{eff} **1975**	+	0,806 ***	+	0,613 ***	-	0,739 ***	-	0,758 ***
	m_{eff} **1968**	+	0,813 ***	+	0,629 ***	-	0,736 ***	-	0,755 ***
	m_{eff} **1930**	+	0,690 ***	+	0,478 *	-	0,733 ***	-	0,674 ***
			N= 24		N= 26		N= 24		N= 26
Wildschwein	m_{eff} **2002**	+	0,553 **	+	0,312 ns	-	0,898 *** 1)	-	0,785 *** 1)
	m_{eff} **1995**	+	0,551 **	+	0,311 ns	-	0,898 *** 1)	-	0,786 *** 1)
Sus scrofa	m_{eff} **1988**	+	0,551 **	+	0,310 ns	-	0,898 *** 1)	-	0,785 *** 1)
	m_{eff} **1975**	+	0,548 **	+	0,297 ns	-	0,904 *** 1)	-	0,797 *** 1)
	m_{eff} **1968**	+	0,548 **	+	0,314 ns	-	0,890 *** 1)	-	0,791 *** 1)
	m_{eff} **1930**	+	0,454 *	+	0,174 ns	-	0,860 *** 1)	-	0,735 *** 1)
			N= 24		N= 25		N= 23		N= 24
Fuchs	m_{eff} **2002**	+	0,701 *** 1)	+	0,681 *** 1)	-	0,613 ** 1)	-	0,541 ** 1)
	m_{eff} **1995**	+	0,703 *** 1)	+	0,682 *** 1)	-	0,614 ** 1)	-	0,540 ** 1)
Vulpes vulpes	m_{eff} **1988**	+	0,705 *** 1)	+	0,683 *** 1)	-	0,618 ** 1)	-	0,537 ** 1)
	m_{eff} **1975**	+	0,710 *** 1)	+	0,688 *** 1)	-	0,619 ** 1)	-	0,544 ** 1)
	m_{eff} **1968**	+	0,714 *** 1)	+	0,695 ***1)	-	0,612 ** 1)	-	0,557 ** 1)
	m_{eff} **1930**	+	0,645 *** 1)	+	0,620 ***1)	-	0,543 ** 1)	-	0,437 * 1)
			N= 24		N= 26		N= 24		N= 26
Dachs	m_{eff} **2002**	+	0,698 *** 1)	+	0,257 ns	-	0,388 ns	-	0,148 ns
	m_{eff} **1995**	+	0,700 *** 1)	+	0,259 ns	-	0,388 ns	-	0,151 ns
Meles meles	m_{eff} **1988**	+	0,698 *** 1)	+	0,253 ns	-	0,386 ns	-	0,148 ns
	m_{eff} **1975**	+	0,705 *** 1)	+	0,241 ns	-	0,370 ns	-	0,147 ns
	m_{eff} **1968**	+	0,715 *** 1)	+	0,243 ns	-	0,356 ns	-	0,159 ns
	m_{eff} **1930**	+	0,583 ** 1)	+	0,011 ns	-	0,356 ns	-	0,003 ns
			N= 23		N= 21		N= 22		N= 20

Tab. 4.2*: Ergebnisse der linearen Regression mit Rehbestand (Jagdstrecke pro ha) bzw. Rehunfallrate (Fallwild/ Jagdstrecke) als abhängige Variablen auf Ebene der hessischen Landkreise. Angegeben sind die Vorzeichen der Regressionskoeffizienten, R als Maß für die Güte der Anpassung und die Signifikanzniveaus (***: p<0,001/ **: p<0,01/ *: p<0,05/ ns= nicht signifikant). Da der Index m_{eff} mit zunehmender Zerschneidung abnimmt, ist der Regressionskoeffizient zum Einfluss von m_{eff} auf den Bestand positiv (bei der Unfallrate umgekehrt).*

		Bestand 2001/02		**Unfallrate 2001/02**
Siedlung [ha/ha]	-	0,892 ***	+	0,804 ***
Dichte Überlandstraßen [km/km²]	-	0,852 ***	+	0,783 ***
meff 2002 [km²]	+	0,805 ***	-	0,725 ***
Wald [ha/ha]	+	0,749 ***	-	0,547 **
Dauergrünland [ha/ha]	+	0,634 ***	-	0,571 **
Landwirtschaft [ha/ha]	+	0,578 **	-	0,638 ***
Brache [ha/ha]	+	0,403 ns	-	0,581 **
Ackerland [ha/ha]	+	0,319 ns	-	0,453 *

Einfache lineare Regressionen mit der Rehunfallrate als abhängige Variable zeigen, dass die Unfallrate mit zunehmendem Siedlungsanteil eines Landkreises steigt (p<0,001), ebenso mit zunehmender Straßendichte (p<0,001) und dem zunehmendem

Grad der Landschaftszerschneidung, der mit m_{eff} gemessen wurde (p<0,001). Die Unfallrate sinkt mit zunehmenden Flächenanteilen von Landwirtschaft (p<0,01), Brache (p<0,01), Dauergrünland (p<0,01) und Waldfläche (p<0,01) (Tab. 4.2).

Eine Hauptkomponentenanalyse sollte klären, welche Zusammenhänge zwischen den Einflussvariablen bestehen. Es wurden alle Variablen in die Analyse einbezogen, die nach den Ergebnissen der Regressionsanalysen den Rehbestand signifikant beeinflussen. Es konnten zwei Hauptkomponenten extrahiert werden (**Tab. 4.3**), die zusammen 98% der Varianz erklären. Der erste Faktor „Landwirtschaftliche Nutzung" erklärt rund 70% der Gesamtvarianz. Hier wirken Landwirtschaftsfläche und Dauergrünland. Der zweite Faktor „Unzerschnittene Waldgebiete", in dem Wald und m_{eff} wirken, erklärt rund 18% der Gesamtvarianz. Siedlungen und Straßendichte, die nahezu identisch wirken, weisen für beide Komponenten Ladungen auf, zeigen aber höhere Ladungen für den Faktor „Landwirtschaftliche Nutzung". Je stärker die landwirtschaftliche Nutzung eines Landkreises, desto dichter ist das Siedlungs- und Verkehrswegenetz. Zudem hängen Siedlungen und Straßen per Definition mit dem Index m_{eff} zusammen. Der Zusammenhang der Variablen untereinander wird auch im Dendrogramm deutlich (**Fig. 4.6**).

Tab. 4.3: *Ergebnisse der Hauptkomponentenanalyse. Komponentenmatrix nach Varimax-Rotation mit Faktorladungen. Komponente 1 „Landwirtschaftliche Nutzung" erklärt 70% der Gesamtvarianz, Komponente 2 „Unzerschnittene Waldgebiete" erklärt 18% der Gesamtvarianz.*

	Komponente	
	1	2
Landwirtschaft	,977	,016
Dauergrünland	,772	,340
Wald	,008	,979
meff 02	,431	,797
Siedlung	-,763	-,618
Strassendichte	-,621	-,721

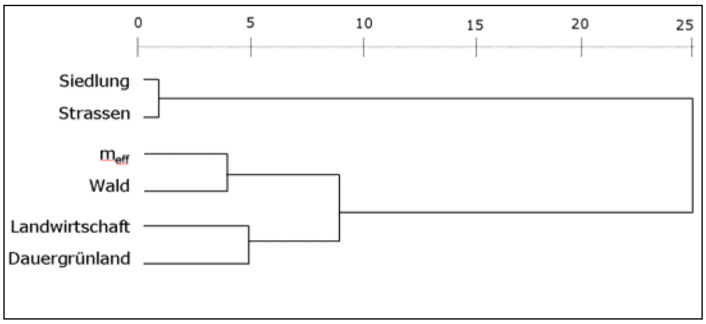


Fig. 4.6*: Dendrogramm für die Effektive Maschenweite (m_{eff}) und die Flächeanteile verschiedener Landnutzungsformen. Die Skala gibt die Größe der Clusterdistanzen im transformierten Wertebereich zwischen 0 und 25 an.*

Eine multiple Regression sollte klären, wie die verschiedenen Einflussvariablen den Rehbestand und die Rehunfallrate gemeinschaftlich beeinflussen. In einem ersten Modelldurchlauf wurden erklärende Variablen vom Statistikprogramm automatisch aufgrund ihres signifikanten Beitrags ausgewählt. Rund 85% der Varianz im Rehbestand können danach durch Siedlungs- und Waldanteil erklärt werden. Rund 65% der Variation der Rehunfallrate können allein durch die Siedlungsfläche erklärt werden (**Tab. 4.4**).

In einem zweiten Modelldurchlauf mit der Rehunfallrate als abhängige Variable wurden von vornherein Siedlungs- und Verkehrsfläche wegen ihrer definitionsgemäß hohen Kollinearität mit m_{eff} aus der Analyse ausgeschlossen und konnten deswegen im Regressionsmodell nicht berücksichtigt werden. In diesem Fall werden aus den verfügbaren Einflussfaktoren m_{eff} und die Landwirtschaftsfläche ausgewählt, die zusammen rund 65% der Varianz in der Rehunfallrate erklären (**Tab. 4.5**).

Die Ergebnisse der Hauptkomponentenanalyse verdeutlichen, warum m_{eff} in den multiplen Regressionsmodellen nicht als erklärende Variable aufgenommen wird. Zunächst weist m_{eff} höhere Faktorladungen für die zweite Komponente auf, die einen schwächeren Teil der Gesamtvarianz erklärt als die erste Komponente (**Tab. 4.3**). In der zweiten Komponente wirkt m_{eff} dagegen schwächer als der Waldanteil, was an der kleineren Faktorladung deutlich wird. Insgesamt bleibt m_{eff} also wegen der hohen Kollinearität der Einflussfaktoren zugunsten aussagestärkerer Variablen in den Regressionsmodellen unberücksichtigt. Lediglich wenn Siedlungen und Straßen von vornherein vom Regressionsmodell ausgeschlossen werden, wird der Index m_{eff} als Variable zur Erklärung der Unfallrate aufgenommen.

***Tab. 4.4:** Schrittweise multiple Regressionen zum Einfluss von Landschaftszerschneidung und Landnutzung auf Bestand und Unfallrate von Rehen. Erster Modelldurchlauf: Die Einflussvariablen wurden von SPSS automatisch ausgewählt aus allen Parametern, die in der einfachen Regression signifikant waren (vgl. **Tab. 4.2**). Modellkennwerte Bestand: R^2_{korr}=0,83, p< 0,001; Unfallrate: R^2_{korr}=0,63, p< 0,001.*

Zielvariable	Prädikatoren	B	Signifikanz
Bestand	Siedlung	-0,083	<0,001
	Wald	-0,033	0,013
Unfallrate	Siedlung	0,014	<0,001

***Tab. 4.5:** Schrittweise multiple Regressionen zum Einfluss von Landschaftszerschneidung und Landnutzung auf die Unfallrate von Rehen. Zweiter Modelldurchlauf: Siedlungen und Strassen werden wegen der hohen Kollinearität mit m_{eff} im Vorfeld vom Anwender aus der Analyse ausgeschlossen. Modellkennwerte R^2_{korr}= 0,62, p< 0,001.*

Zielvariable	Prädikatoren	B	Signifikanz
Unfallrate	meff 02	-0,013	<0,001
	Landwirtschaft	-0,005	0,011

4.4 DISKUSSION

Die effektive Maschenweite (m_{eff}) zur Messung der Landschaftszerschneidung erfreut sich in jüngster Vergangenheit großer Popularität und hat sich längst aus der rein wissenschaftstheoretischen Anwendung gelöst und Eingang in Umweltmonitoringsysteme und die angewandte Naturschutzpolitik gefunden. Inzwischen dient m_{eff} den Umweltbehörden verschiedener Bundesländer als Messgröße, um den Zustand und die Entwicklung der Flächenzerschneidung zu dokumentieren (LfUG 2002, Esswein et al. 2002, 2004a, 2004b, Neumann-Finke 2004, Voerkel 2005, Roedenbeck et al. 2005).

Die Länderinitiative Kernindikatoren (LIKI) empfiehlt, den Index zukünftig in einer indikatorbasierten Umweltbewertung mit erhöhter Priorität zu behandeln (Schupp 2005). Vor dem Hintergrund dieser Entwicklung scheint es wichtig zu überprüfen, ob die Indikationsfähigkeit von m_{eff} auf die Quantifizierung der Landschaftszerschneidung beschränkt ist, oder ob der Index auch eine qualitative Aussage treffen kann - z.B. zur Überlebensfähigkeit von Tierpopulationen in zerschnittenen Lebensräumen.

Die vorliegende Arbeit zeigt, dass sich eine Analyse zur Beantwortung dieser Fragestellung großen Herausforderungen stellen muss. Welche Aussagekraft die vorliegenden Ergebnisse besitzen und welche Interpretationsschwierigkeiten auf Landschaftsebene bestehen soll hier diskutiert werden.

4.4.1 Zur Verwendung der Streckenlisten

Ein generelles Problem bei Analysen über landschaftliche Effekte von Straßen ist die Auswahl der Tierbestandsdaten. Mit zunehmender Größe des Untersuchungsgebiets steigt zwangsweise die Anzahl der beteiligten Datenerfasser. Deren Qualifikation und die Kartierintensität variieren individuell und lokal und beeinflussen die Datenqualität. Hinzu kommt das Problem, dass bei großflächigen Kartierungen meist keine Flächendeckung gegeben ist.

Die für diese Analyse verwendeten Streckenlisten zeigen eine hohe Flächendeckung, denn sie sind von der Revier- über die Landkreisebene bis zum Bundesland verfügbar. Allerdings stellen sie nur eine Annährung an die tatsächliche Bestandssituation dar (Strauss 2000). Für Rehe ist davon auszugehen, dass die Abschusszahlen als relativ guter Bestandsindikator herangezogen werden können, denn für diese Art werden jährlich Abschusspläne erstellt, die auf Gutachten zu Verbissschäden basieren (Scheffler 2005, mündl. Korrespondenz). Dabei wird davon ausgegangen, dass die Verbissstärke in Relation zum Bestand steht und der festgesetzte Abschuss sich entsprechend am Bestand orientiert – er ist für die Jagdrevierinhaber verpflichtend. Bei den anderen Arten ist diskussionswürdig, ob sich die Streckenlisten als Bestandsangaben eignen. Durch regional unterschiedliches Jagdinteresse oder regionale Schonprogramme ist eine zeitlich und lokal konstante Bejagungsintensität nicht immer gegeben und die unterschiedliche Bejagungsaktivität kann zu einer Fehleinschätzung der realen Abundanzen führen, so wie es für Niederwildarten derzeit diskutiert wird (Strauss 2000, Strauss und Pohlmeyer 2001).

Eine unabdingbare Voraussetzung für jedwede Untersuchung auf Landschaftsebene wäre die Einrichtung landesweit flächendeckender Monitoringprogramme. Auf einer entsprechenden Grundlage könnte die vorliegende Analyse auch für naturschutzrelevante Arten erfolgen und hätte sicher höhere Aussagekraft. Es ist bemerkenswert, dass landschaftsökologische Erkenntnisse für die Entscheidungsfindung in der Straßenplanung dringend benötigt werden, während einwandfreie, landesweite Datengrundlagen für eine entsprechende Analyse nicht verfügbar sind. Die Datenqualität gilt es also bei der Interpretation der Ergebnisse zu berücksichtigen.

4.4.2 Zur Interpretation der Bestandsentwicklung

Für keine der vier untersuchten Arten sind in den letzten Jahren in Hessen Bestandsverluste zu verzeichnen. Trotzdem eignen sich die Arten für eine landschaftsökologische Untersuchung zu den Effekten der Landschaftszerschneidung. Diese vier letzten Großwildarten mit annährend flächendeckendem Vorkommen in Hessen gelten als Stellvertreter für andere Wildtiere mit ähnlich großen Raum- und Habitatansprüchen. Für naturschutzfachlich relevantere Arten, wie Luchs und Wildkatze, ist aufgrund ihres lückenhaften Vorkommens eine empirische Analyse nicht möglich. Hier ermöglichen ausschließlich Modellsimulationen Aussagen zu den Effekten der Landschaftszerschneidung (Klar et al. 2006).

Für die Bestandszunahmen der untersuchten Arten werden in der Literatur unterschiedliche Einflussfaktoren diskutiert. Als Gründe für den rasanten Anstieg in der Wildschweinstrecke wird beispielsweise die veränderte Anbaustruktur in der Landwirtschaft genannt, häufige Mastjahre, milde, schneearme Winter und Veränderungen im Waldaufbau. Die unverhältnismäßig hohe Verabreichung von Futtermitteln und die artbedingt hohe Vermehrungsrate haben dazu geführt, dass die Populationen allgemein als zu hoch mit Tendenzen zur weiteren Ausbreitung erachtet werden (Bartel et al. 2005). Auch die Dachsstrecke hat nach einem starken Rückgang in den 50er Jahren, deren primäre Ursache das Tollwutgeschehen war, nach dem Zurückdrängen der Viruserkrankung und einer zurückhaltenden Bejagung in Hessen wieder nachhaltig zugenommen. Ohne dass an dieser Stelle eine detaillierte, wildtierbiologische Betrachtung für alle Arten erfolgen kann, wird zusammenfassend festgehalten, dass in den vergangenen 70 Jahren wesentliche Veränderungen im politischen, sozioökonomischen und ökologischen Bereich stattgefunden haben. Diese Faktoren haben Bestandsentwicklungen wesentlich mitgeprägt, überlagern den Einfluss der Landschaftszerschneidung und erschweren eine klare Kausalitätsanalyse. Die Interpretation von Zeitreihen ist deswegen wenig aussagekräftig. Um den Einfluss anderer Landschaftsveränderungen ausschließen zu können, muss eine Analyse räumlich differenziert erfolgen.

Eine Schlussfolgerung lässt die Zeitreihenanalyse allerdings zu: Neben den positiven Bestandsentwicklungen ist zu beobachten, dass im Zeitverlauf die Rate der im Verkehr getöteten Wildtiere unverhältnismäßig stark angestiegen ist. Erhöhte Unfallhäufigkeiten sind zwar primär auf erhöhte Bestände zurückzuführen, werden darüber hinaus aber eindeutig von verkehrsabhängigen Faktoren beeinflusst. Beim Reh nehmen die Fallwildzahlen rund 20% der Gesamtstrecke ein. Wildunfälle mit Rehen und Wildschweinen sind auch für den Menschen gefährlich und verursachen jährlich einen Versicherungsschaden von rund 50 Mio. Euro in Deutschland. Beim Dachs können die Verkehrsverluste mit 30-49% der Gesamtstrecke zweifelsohne als bestandsgefährdend eingestuft werden.

Auf Landkreisebene wird deutlich, dass räumliche Muster von Wildtierbeständen und Unfallraten mit dem Grad der Landschaftszerschneidung zu erklären sind. Bei den Arten Reh, Wildschwein, Fuchs und Dachs ist eine hohe Verkehrsnetz- und Siedlungsdichte mit geringeren Beständen korreliert. Bei Reh, Wildschwein und Fuchs, steigt der Anteil am Bestand, der durch Verkehrsunfälle abgeschöpft wird, mit zunehmender Landschaftszerschneidung. Es ist zu vermuten, dass die Wildtierarten mit Zeitverzögerung auf die zunehmende Landschaftszerschneidung reagieren, denn die aktuellen Bestände sind am Besten an die Zerschneidungssituation von vor rund 30 Jahren angepasst. Zur Aussagekraft der hier gefundenen Ergebnisse muss die Frage beantwortet werden, welche Bedeutung dem Zerschneidungsgrad im Verhältnis zu anderen Einflussfaktoren zukommt. Auf Grundlage von Regressionsanalysen lassen sich folgende Schlüsse zur Indikationsqualität von m_{eff} ziehen:

(1) In multiplen Regressionsanalysen wird m_{eff} wegen der hohen Kollinearität der Einflussfaktoren aus den Modellen ausgeschlossen. Die reine Straßendichte und die Flächenanteile von Siedlungen und Wald sind offensichtlich besser geeignet, um Bestandsdichten und Unfallraten von Rehen zu erklären.

(2) In einfachen Regressionsanalysen wird deutlich, dass m_{eff} neben dem Flächenanteil von Siedlung und Verkehrsfläche die drittwichtigste Einflussvariable für Rehbestand und Unfallrate ist. Neben den reinen Flächenanteilen spielt offensichtlich auch die räumliche Konfiguration des Siedlungs- und Verkehrswegenetzes eine Rolle, die in dem Index ihren Ausdruck findet.

(3) Schwellenwerte der Landschaftszerschneidung, bei denen Wildtiere mit überdurchschnittlichen Bestandseinbußen reagieren, wären mit Hilfe der Effektiven Maschenweite höchstwahrscheinlich messbar, wenn größere Stichprobenumfänge gegeben wären.

(4) Die Bedeutung des Indices m_{eff} geht über eine reine Quantifizierung der Landschaftszerschneidung durch Siedlungen und Straßen hinaus, da der Index mit dem Waldanteil eines Landkreises korreliert ist und somit eine zusätzliche Aussage zu einem Habitatparameter („unzerschnittene Waldgebiete“) trifft. Für waldgebundene Arten mit großen Raumansprüchen ist der Index damit grundsätzlich in der Lage, Bestandsdichten und Unfallraten anzuzeigen. Bei nicht-waldgebundenen Arten wäre die Indikationsqualität noch zu prüfen und ist vor dem Hintergrund der hier gefundenen Ergebnisse zumindest fraglich.

4.5 SCHLUSSFOLGERUNGEN UND AUSBLICK

Die vorgestellten Ergebnisse zeigen, dass eine Analyse auf Landschaftsebene oft zu Interpretationsschwierigkeiten führt. Problemfelder sind (1) die mangelnde Verfügbarkeit flächendeckender Wildtierbestandsdaten, (2) historische Landschaftsveränderungen, die neben der Landschaftszerschneidung Bestandsentwicklungen mit beeinflussen und (3) die Kollinearität verschiedener Einflussparameter in realen, hochdiversen Landschaften.

Trotz dieser Einwände bleibt bestehen, dass die effektive Maschenweite ein sinnvoller Index ist, um die Landschaftszerschneidung zu messen. Der Index kann zeigen, dass die Landschaftszerschneidung in Hessen seit 1930 zugenommen hat, obwohl politische Entscheidungsträger seit langem eine „Trendwende im Landverbrauch" fordern (Bundesminister des Inneren 1986). Der Index m_{eff} konzentriert sich nicht - wie zum Beispiel die UZR des BFN - ausschließlich auf die Flächengröße der größten Landschaftsräume, sondern bezieht auch kleine Räume und deren räumliche Konfiguration ein. Für waldgebundene Arten mit eher generalistischen Standort- und großen Raumansprüchen, die in ihrer Fortbewegung permanent mit der Barriere- und Mortalitätswirkung von Straßen konfrontiert sind, trifft m_{eff} Aussagen, die über eine reine Quantifizierung der Zerschneidung hinausgehen.

Da die Landschaftszerschneidung offensichtlich negative Auswirkungen auf Wildtierpopulationen hat, lassen sich folgende Empfehlungen ableiten:

(1) Wildunfälle haben über die letzten Jahre unverhältnismäßig zur Bestandsentwicklung zugenommen. Zur Erhöhung der Verkehrssicherheit und zum Schutz von Wildtieren müssen Querungshilfen in zerschneidungsintensiven Regionen als Minimierungsstrategie diskutiert werden.

(2) Die Bestandsgrößen von vier Wildtierarten mit großen Raumansprüchen sinken durch den Verlust großer, unzerschnittener Waldgebiete. Solche Gebiete sind eine wichtige Ressource für Arten mit großen Raumansprüchen und als Erholungsraum für den Menschen und verdienen erhöhte Beachtung im Gebietsschutz.

(3) Bei straßenbaulichen Maßnahmen ist nicht nur die Beeinträchtigung des angrenzenden Habitats zu bewerten und auszugleichen, sondern insbesondere die Aufrechterhaltung funktionaler Beziehungen auf Landschaftsebene.

Generell ist allerdings davor zu warnen, große unzerschnittene Räume in der Umweltplanung als „Schutzgut" über die bisher in der Planungsmethodik berücksichtigten Schutzgüter (hier insbesondere Arten und Lebensräume) zu stellen. Die Landschaftszerschneidung, die mit m_{eff} quantifizierbar ist, kann nicht mit Habitatzerschneidung gleichgesetzt werden. Die Lebensräume vieler, insbesondere nicht-waldgebundener Arten orientieren sich an den abiotischen Voraussetzungen und dem Habitatangebot. Habitate liegen sozusagen in „Subsystemen" innerhalb unzerschnittener Räume und überspannen diese auch. Auch in kleinen Landschaftsräumen kommen gefährdete Arten vor, die an spezielle Standortbedingungen gebunden sind. Für Habitatspezialisten ist m_{eff} sicher kein guter Index zur Bewertung der Überlebensfähigkeit von Populationen. Große unzerschnittene Räume erfüllen erst neben kleinen Trittsteinbiotopen hoher Habitatqualität die Anforderungen eines landschaftlich wirksamen Biotopverbunds. Entscheidungen bei Eingriffsvorhaben nur nach dem Kriterium „Erhalt großer unzerschnittener Räume“ zu treffen wäre fatal, wenn dadurch Lebensräume schutzbedürftiger Arten ausgelöscht werden würden. Bei eingriffsbezogenen Umweltplanungen (z.B. UVP) ist demnach der Umgang mit qualitativen Schutzgütern geboten. Es wird empfohlen, die Effektive Maschenweite nicht als alleiniges Kriterium zur Bewertung der Zerschneidungssituation zu verwenden, sondern parallel zielartenspezifische Qualitätsindikatoren im Umweltmonitoring einzusetzen.

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Chapter 5.
Effects of roads on spatial distribution, abundance, and road mortality of brown hare (*Lepus europaeus*) in Switzerland

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Abstract

European hare populations (Lepus europaeus) are in decline since the 60ies and numerous impact factors have been discussed in literature. Although landscape fragmentation by roads is assumed to be one potential factor, the effects of roads on hare populations are barely understood. We studied three potential road effects asking: (1) Do roads affect spatial distribution of hares due to disturbance effects? (2) Does road network density affect hare abundance due to barrier effects? (3) Does road network density affect road mortality rates? The study is based on harvest statistics and spotlight taxations in Canton Aargau, Switzerland, and was conducted at three different spatial scales. Spatial distribution was studied in plots established in varying distances parallel to roads, effects on abundance were analysed on the basis of raster grids, and road mortality was studied on hunting district level.

We show that (1) hares avoid the proximity to roads, and prefer large non-fragmented areas in contrast to small isolated patches. (2) The density of highways, federal and main roads has a negative effect on hare abundance. The density of unpaved field tracks has a positive effect, probably because vegetation at field tracks matters in the diet spectrum. (3) Effects of road network density on road mortality rates could not be shown, although road mortality increases since the 90s. We conclude that in a multi-factorial complex of impact factors, including weather conditions, diseases, predation, hunting and habitat quality, roads are of secondary importance. However, in debilitated populations roads act as additive threatening factor.

Zusammenfassung

Effekte von Straßen auf die Raumnutzung, Bestandsdichte und Verkehrsmortalität von Feldhasen (Leous europaeus) in der Schweiz. *Populationen des Feldhasen (Lepus europaeus) sind seit den 60er Jahren im Rückgang inbegriffen und eine Vielzahl möglicher Ursachen wird in der Literatur diskutiert. Obgleich die durch Straßen verursachte Landschaftszerschneidung als ein potentieller Einflussfaktor diskutiert wird, sind die Effekte von Straßen auf Feldhasenpopulationen bislang immer noch weitestgehend unbekannt. In der vorliegenden Studie wurden drei potentielle Straßeneffekte untersucht: (1) Beeinflussen Straßen die Raumnutzung von Feldhasen aufgrund von Störungseffekten? (3) Beeinflusst die Dichte von Straßennetzen die Populationsdichte von Feldhasen aufgrund von Barriereeffekten? (3) Beeinflusst die Dichte von Straßennetzen die Verkehrsmortalitätsrate? Die Studie basiert auf Jagdstreckendaten und Scheinwerfer-Taxationen im schweizerischen Kanton Aargau und wurde auf drei unterschiedlichen Raumebenen durchgeführt. Die Raumnutzung wurde in Plots untersucht, eingerichtet in unterschiedlichen Entfernungen parallel zur Straßen. Effekte auf Populationsdichten wurden auf der Basis eines Rastergitters, die Verkehrsmortalität auf der Basis von Jagdrevieren untersucht.*

Es wird gezeigt, dass (1) Feldhasen die Nähe zu Straßen meiden und große unzerschnittene Gebiete gegenüber kleinen isolierten Flächen bevorzugen. (2) Die Dichte von Autobahnen, Bundes- und Hauptstraßen hat einen negativen Effekt auf Feldhasendichten. Die Dichte von Feldwegen hat dagegen einen positiven Effekt, wahrscheinlich weil die Vegetation an Feldwegen eine Rolle im Nahrungsspektrum spielt. (3) Es konnten keine Effekte der Straßendichte auf Verkehrsmortalitätsraten gezeigt werden, obwohl die Verkehrsmortalität seit den 90er Jahren ansteigt. In einem Multifaktorenkomplex verschiedener Einfluss-größen, einschließlich Wetterbedingungen, Seuchen, Fressfeinden, Jagddruck und Habitatqualität sind Straßen scheinbar von sekundärer Bedeutung. In geschwächten Populationen wirken sie allerdings als additiver Gefährdungsfaktor.

5.1 INTRODUCTION

European hare (*Lepus europaeus*) populations are in decline throughout Europe since the 1960ies (Ninov 1990, Pielowski 1990, Smith et al. 2005). Many authors hypothesize that the decline observed is due to intensification of agriculture leading to field enlargement and block farming (Edwards et al. 2000, Lewandowski and Nowarowski 1993, Lundström-Gilliéron and Schlaepfer 2003, Schröpfer and Nyenhuis 1982, Tapper and Barnes 1986). In the new states of Germany, for instance, the decline is possibly due to the structural change in agriculture caused by the German reunification (Ahrens and Kottwitz 1997). Competition pressure of predator populations (Reynolds 1995), hunting and weather conditions (Smith et al. 2005) are expected to be other factors causing population declines. As a consequence of population development, hare is listened as a threatened species in Switzerland today (Pfister et al. 2002).

Above all, landscape fragmentation by roads and traffic is expected to be another threatening factor (Lundström-Gilliéron and Schlaepfer 2003). There are three main effects potentially arising from roads on hare populations. (1) *Disturbance effects* arising from vehicles and traffic may lead to avoidance of the area adjacent to roads, and to changes in movement and behavioural patterns. (2) *Barrier effects* may arise, when the exchange of individuals between habitats is hindered by highly frequented roads due to disturbance effects or road mortality. As a consequence of limited exchange, genetic variability may decline, affecting abundances and population persistence. (3) R*oad mortality* is often presumed to be the main effect. In Czechia and in Austria, for instance, brown hare is one of the species being most often killed by traffic (Glitzner et al. 1999).

Despite a fairly large number of population vulnerability analyses studying population survivability of brown hare in the context of multiple impact factors (review in Smith et al. 2005), the exclusive effects of roads have barely been studied. Hence, the potential road effects discussed above are not proven yet.

Concerning distribution of brown hare in space first observations in Northern Germany indicate that the abundance of hares is smaller in areas adjacent to roads than further away in the field (Strauß and Pohlmeyer 2001). However, these observations have not been studied systematically and there is no study proving these assumptions.

Concerning barrier effects results of previous studies are contradictory. It has been shown in one study that the genetic structure of brown hare populations differed significantly in two populations inhabiting two areas separated by a highway (Fickel et al. 1999). However, in another study in Austria the presence of highways explained none of the allelic differentiation among populations of brown hares (Hartl et al. 1989). Population vulnerability analysis including roads as a potential impact factor could not find a relationship between road density and hare abundance (Hoffmann 2003). It is unclear thus far, whether roads actually cause barrier effects, and whether these barrier effects truly affect population abundance.

Concerning road mortality we do not know, whether road mortality indeed affects population persistence. As none of the road mortality studies on brown hare conducted thus far collected information on population size, it is conceivable that losses of individuals are compensated by mechanisms and population dynamics such as reductions in other mortality sources or increases in reproduction.

The objective of our study was to analyse whether and in which ways roads affect hare populations. Following the three main road effects discussed above we ask: (1) Do roads affect spatial distribution of hares due to disturbance effects? (2) Does road network density affect hare abundance due to barrier effects? (3) Does road network density affect road mortality rates of hares? The particularity of our study is that it is not a classical population vulnerability analysis (e.g. Hoffmann 2003, Pegel 1986). In fact, we intended to exclude other potential impact factors from the study design to analyse the exclusive effect of roads.

5.2 MATERIAL AND METHODS

5.2.1 STUDY AREA

Study area was Canton Aargau in Northern Switzerland with a topography ranging from the river Rhine to low mountain ranges and colline zones in the Jurassic (max. 908m ASL) (**Fig. 5.1**).

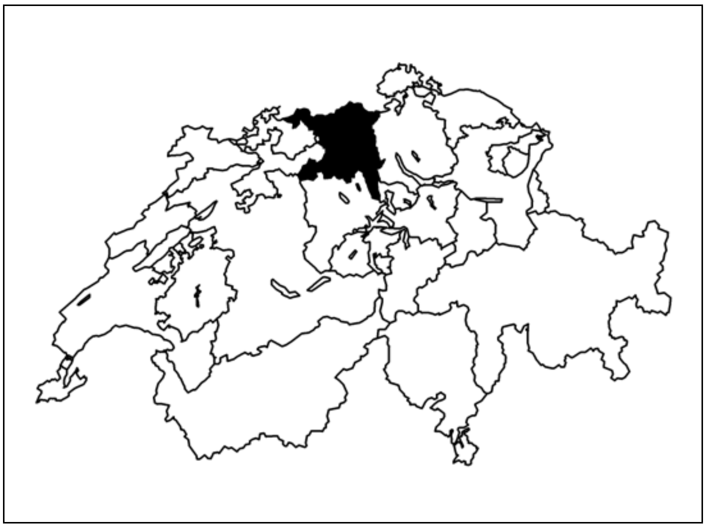

***Fig 5. 1**: Study area Canton Aargau in Northern Switzerland (1,404 km²).*

With a total area of 1,404 km², 45% of the Canton is intensively cultivated land used by agriculture, 15% is urban area, and 37% is used by forestry with 42% of the forest area consisting of beech groves, 37% mixed forests, and 21% spruce plantations in the year 2005. In the triangle between Zurich, Bern and Basel Canton Aargau is an important economic factor and a hub in the traffic network of Switzerland. With a road network of 1,150km of cantonal roads, 110 km of national roads, and 5km of highways mobility is high, and the area occupied by traffic infrastructure is larger than in the Swiss average. Average daily traffic increased constantly since 1950 and reached 140,000 cars per day in the year 2001.

5.2.2 DATA

Geographical data of roads, water courses, land-use (forest, agricultural fields, orchards, fallow land and urban areas) and natural landscapes (e.g. hedges, nature conservation areas, ecological buffer areas, natural grasslands), were obtained from the AGIS-datapool of the cantonal administration of Aargau. Data were provided as GIS-Shapefiles. Landscape topography was obtained from a digital elevation database (DHM25 matrix model) with a spatial resolution of 25m² cells.

Road kill data were obtained from hunting statistics maintained by about 1,200 hunters on the level of 219 hunting districts. Hunting statistics include numbers of hares hunted, and numbers of hares found dead due to other reasons than hunting, which might be road kills, diseases, dogs, pesticides and age. However, road mortality is by far the most common factor, so we will refer to this as the number of road kills below. Harvest numbers were available for the years 1933-2005, road kill data for the years 1971-2005.

Population abundances were obtained from hare taxations conducted by huntsmen in the years 2003 and 2005 by order of the cantonal administration. In these taxations hares were counted in all hunting districts by means of spotlight taxation – an approved and standardised methodology used to obtain hare abundances (Strauß and Pohlmeyer 1997). For the taxation a vehicle drove along all field tracks in a hunting district following a fixed route. Along the way the whole field area was illuminated by a spotlight fixed at the vehicle perpendicular to the driving direction. All hares in the cone of light visible to the naked eye were counted and plotted into topographical maps (1:12,500). All hares resident in forest edges usually move into the fields during the night and could be counted the same way, although there was no mapping in forest areas. Each year the spotlight taxation was repeated in two nights in February or March, and exceptionally in April (overall 4 taxations: 2 years x 2 mappings). All data were collected by the cantonal administration and digitized with a Geographical Informationing System (GIS).

Landscape analysis such as point digitalisation and parameter calculation were performed using Geographical Information System software ArcView 3.2 and ArcGIS 8.0 (ESRI). Statistical analyses were performed using SPSS 11.5.1.

5.2.3 ANALYSES OF HABITAT PREFERENCES

We first conducted an analysis figuring out habitat preferences of hares. Results of this study were used in further analysis to exclude habitat as a potential impact factor on spatial distribution, population abundance, and road mortality. We analysed habitat preferences by comparing hare locations with randomly selected control locations.

Data base for hare locations was the spotlight taxation of the year 2005. Control sites were randomly set by the GIS-Tool *Random Point Generator* (Jenness Enterprises 2005) on the entire agricultural field of the study area.

We calculated 17 different variables to describe landscape characteristics in close proximity to hare and control locations (**Tab. 5.1**). These variables describe the fragmentation and isolation situation, as well as habitat and resource availability, the amount of ecologically valuable areas and neighbourhood effects. For calculation of some variables we established a buffer (150m) around each hare and control location. Only points with buffers covering the whole study area were used for further analyses (N= 2,413 hare sites; N= 2,436 control sites).

To check the differences between hare and control locations we compared the estimates of metric variables by non-parametric Mann-Whitney-U-Test. Then we separated the data into two subsets, and used the first spotlight taxation of 2005 for model building (N= 1,444 hare sites; N=1,446 control sites), and the second spotlight taxation of 2005 for model validation (N=993 hare sites; N=993 control sites). All variables distinguishing significantly between hare and control locations following the results of U-Test were included into model building. We generated predictive models by generalized linear models using binominal logistic regression analysis, with site status as binary response variable (hare: 1, control: 0). The final parameter combination was generated by forwards selection method.

***Tab. 5.1**: Landscape and fragmentation variables calculated to analyse habitat preferences of brown hare in Canton Aargau (Switzerland). Variables were measured in a 150m buffer around 2,413 hare locations and 2,436 randomly distributed control locations.*

Fragmentation parameters		
Parameter	**Unit**	**Definition**
$ROAD12_{LENGTH}$	[m]	Length of highways, federal, main and side roads (category 1, 2, and highways).
$ROAD12_{DIST}$	[m]	Distance to next highway, federal, main and side road (category 1, 2 and highways).
$ROAD456_{LENGTH}$	[m]	Length of drivable farm track and unpaved foot ways (category 4, 5 and 6).
$ROAD456_{DIST}$	[m]	Distance to next drivable farm track and unpaved foot way (category 4, 5 and 6).
$PATCH_{AREA}$	[ha]	Area size of non-fragmented patch. Barrier elements: All roads category 1-6 and highways.
Habitat parameters		
Parameter	**Unit**	**Definition**
$FOREST_{DIST}$	[m]	Distance to next forest edge.
$AGRI_{AREA}$	[ha]	Amount of agricultural land per buffer area.
$URBAN_{DIST}$	[m]	Distance to next settlement.
$EDGE_{AREA}$	[ha]	Amount of forest edge per buffer area. This is the area of agricultural fields in a 100m buffer next to forest edges
ECO_{DIST}	[m]	Distance to next ecological buffer area.
$FALLOW_{DIST}$	[m]	Distance to next fallow land.
$GRASS_{DIST}$	[m]	Distance to next semi-natural grassland.
$ORCHARD_{DIST}$	[km]	Distance to next orchard.
$HEDGE_{DIST}$	[m]	Distance to next hedge or solitaire tree.
$WATER_{LENGTH}$	[m]	Length of water courses per buffer area.
ASL_{RANGE}	[m]	Variation in topography, distance between highest and lowest elevation (height above sea level) in buffer.
ASL_{MEAN}	[m]	Mean elevation (height above sea level) in buffer.
MSI	-	Mean shape index of unfragmented patch. Index > 1. (1: round).
$NEIGHBOUR_{DIST}$	[m]	Distance from each hare to next hare, and from each control point to next control point.

Models were validated with 993 hare and 933 control locations not included in the model building process. Values of these points were entered into the model equations to see whether each location was a hare or control point. A location was classified as ´hare site´ if its predicted probability was greater than 50% and a control site if the predicted probability was less than 50% (Bühl & Zöfel 2005). We compared actual states and predictions in 2 x 2 tables. To determine how the final model performed in distinguishing between hare and control locations, univariate logistic regression analysis was used with the actual status of the sites as a binary response variable and the site-specific probability being a hare site as an independent predictor

5.2.4 ANALYSES OF ROAD EFFECTS

Studies analysing road effects were conducted at three different spatial scales (**Fig. 5.2**): (1) disturbance effects with spatial distribution of hares as endpoint were analysed on the level of plots established parallel to roads, (2) barrier effects with hare abundance as endpoint were analysed on raster grid level, and (3) road mortality was analysed on hunting district level.

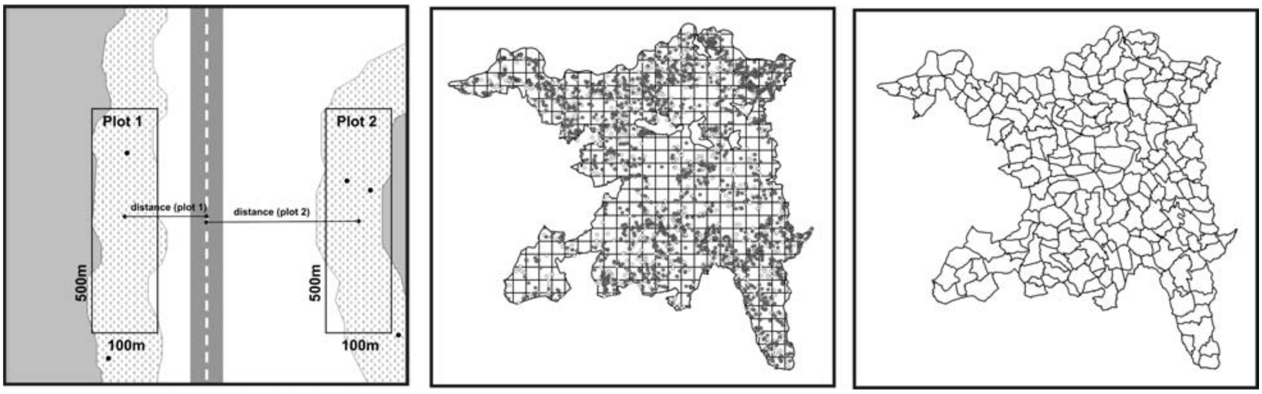

***Fig. 5.2**: Three levels of spatial analysis: (1) Disturbance effects of roads on spatial distribution of brown hare were analysed in plots (500 x 100m) in varying distances parallel to roads, (2) barrier effects on brown hare abundance were analysed at larger scales on raster grid level (2x2 and 4x4 km), and (3) effects on road mortality were analysed on the basis of hunting districts.*

DISTURBANCE EFFECTS

To study disturbance effects of roads we established 111 longitudinal plots (500 x 100m) parallel to relatively straight 500-m road segments in the whole study area (**Fig. 5.2**). Plots were established exclusively next to federal and main roads (road class 1 and 2) to control for traffic densities to some degree. We controlled for habitat parameters by establishing plots only in forest edges. Forest edge area was defined as the agricultural area in a 0-100m proximity to forest edges. This habitat type turned out to be most important for spatial distribution of hares in the previous analysis. Because forest edge habitat did not always cover the whole plot, we calculated the amount of forest edge per plot. All plots were situated in a distance from 0 to 700m to the road.

We measured distance from road centre to plot centre and classified all plots into 5 distance classes (N_0=32; N_{100}=27; N_{200}=18; N_{300}=12; $N_{400-700}$=22).

We calculated number of hares per edge area in each plot based on spotlight taxations in 2003 and 2005, and in each year we used the mapping where more hares had been counted. Finally, we calculated mean density of hares and standard deviation for each density class. We conducted analysis of variance (ANOVA) to test the hypothesis that the means of hare densities are equal among distance classes.

BARRIER EFFECTS

To study barrier effects at larger scales we produced two raster grids covering the whole Canton with a mesh size of 2x2 km and 4x4 km. We cut out all hunting districts from the raster grid where no spotlight taxation had been prosecuted. As these were different districts in all taxations the sample size of raster grid cells varied. We only used grid cells completely covering the Canton. To control for habitat availability we calculated the area of forest edges (see above) per raster grid cell, and only used grid cells with 20-30% forest edge area for further analysis ($N_{4x4/05}$=14; $N_{4x4/03}$=17; $N_{2x2/05}$=76; $N_{2x2/03}$=79).

We counted hare numbers in each complete raster grid cell based on the spotlight taxations of 2003 and 2005, and in each year we used the mapping where more hares had been counted. Then, we calculated density of road networks per raster grid cell. The analysis was conducted for two different road categories: (i) roads (highways, federal, main and side roads; i.e. road class 1 and 2), and (ii) tracks (drivable farm tracks and unpaved footways; i.e. road class 4, 5 and 6) (**Tab. 5.2**). To uncover potential relationships between road network density and hare abundances, respectively, we conducted linear regression analysis.

Tab. 5.2: *Fragmentation and habitat variables calculated on raster grid (2x2 and 4x4 km) and hunting district level to analyse effects of roads on abundance and road mortality of brown hare in Canton Aargau (Switzerland).*

Parameter	Unit	Definition
$ROAD12_{DENS}$	[km/km²]	Road density per hunting district/ grid cell. Highways, federal, main and side roads (category 1, 2, and highways).
$ROAD456_{DENS}$	[km/km²]	Road density per hunting district/ grid cell. Drivable farm tracks and unpaved footways (category 4, 5 and 6).
$EDGE_{PROP}$	[%]	Proportion of forest edge per hunting district. This is the proportion of agricultural area in a 100m proximity buffer close to forests.

ROAD MORTALITY

To study road mortality we conducted an analysis on the basis of hunting districts, because mortality data was based on hunting statistics which were only available on district level. Likewise as for the grid based analysis (see above), we calculated the area of forest edge per hunting district, and only used districts with 20-30% forest edge area to control for habitat availability (N=68). Then, we measured density of (i) roads (highways, federal roads, main and side roads; i.e. road class 1 and 2), and (ii) tracks (drivable farm tracks and unpaved footways; i.e. road class 4, 5 and 6) per hunting district (**Tab. 5.2**). Again, we conducted linear regression analysis to uncover potential relationships between road density and road mortality of hares.

5.3 Results

5.3.1 Habitat preferences

Mann-Whitney U-Test revealed differences between the means of hare and control sites for most of the landscape variables observed (**Tab. 5.3**). Hares primarily preferred the proximity to forest edges ($FOREST_{DIST}$, $EDGE_{AREA}$). Furthermore, hare locations were in closer proximity to ecological buffer areas (ECO_{DIST}), semi-natural grasslands ($GRASS_{DIST}$) and orchards ($ORCHARD_{DIST}$). Hares preferred areas on higher levels (ASL_{MEAN}) with a stronger variation in topography (ASL_{RANGE}). Neighbourhood effects seem to be important as well, as hares preferred the proximity to other hares ($NEIGHBOUR_{DIST}$). There was also a weak preference for fallow land, but this difference was not significant.

On the other hand, hares avoided settlements ($URBAN_{DIST}$), and they seem to avoid hedges ($HEDGE_{DIST}$). Furthermore, hare locations were characterised by a smaller amount of field area in a 150m buffer ($AGRI_{AREA}$). This is because hares prefer forest edges (see above) to areas in the centre of the field.

Concerning landscape fragmentation hares generally preferred large non-fragmented areas ($PATCH_{AREA}$) to smaller patches isolated by traffic infrastructure. The effects of roads on spatial distribution are dependent on road categories. Highways, federal roads, main and side roads are avoided. The distance from hare locations to these roads averages 506.2m in contrast to 369.4m for the control locations ($p<0.001$) ($ROAD12_{DIST}$). In addition to this, hares avoid areas with a dense road network ($ROAD12_{LENGTH}$). The average distance between tracks (drivable farm tracks and unpaved footways) and hare locations (61.9m) is slightly larger than the difference between tracks and control locations (58.5m) ($ROAD456_{DIST}$). However, hares seem to prefer areas with a higher density of farm tracks and footways ($ROAD456_{LENGTH}$).

The final regression model, based on hare abundances of the first spotlight taxation 2005, included two road and four other parameters, and correctly classified 64.8% of hare and control locations. As per this model hares preferred the proximity to forest edges, fellows, farm tracks, footways, and orchards. Settlements and roads (highways, federal, main and side roads) were avoided (**Tab. 5.4**).

When applying the final model to the second spotlight taxation, it correctly classified 713 of 993 hare locations (71.8%), but misclassified 280 hare locations (28.2%) as control sites. From o total of 993 control sites 615 (61.9%) were correctly classified as control locations, and 378 (38.1%) were misclassified as hare locations. Logistic regression analysis with actual status of locations as binary response variable and the site-specific probability as independent predictor was highly significant (Nagelkerke R^2: 14.8%, p<0.001). The model produced an overall concordance of 66.9%, and thus, succeeded in distinguishing between hare and control locations.

Tab. 5.3: *Descriptive statistics for variables determining presence or absence of brown hares. Mean, maximum values and standard error of variables are shown along with results of Mann-Whitney U-Test. Variable coding is explained in **Tab. 5.1**. N= 2,413 hare sites and 2,436 control sites.*

Fragmentation parameters	direction	unit	control (Code 0) max	control (Code 0) mean	control (Code 0) SE	hare (Code 1) max	hare (Code 1) mean	hare (Code 1) SE	U-test Z	U-test P-value
$RD12_{DIST}$	+	[m]	1986.26	369.36	310.33	1951.64	506.24	347.33	-15.647	<0.001
$RD12_{LENTGH}$	-	[m]	1275.86	77.22	143.41	780.60	32.80	92.18	-12.950	<0.001
$RD456_{DIST}$	+	[m]	282.91	58.75	45.59	315.05	61.93	39.42	-5.301	<0.001
$RD456_{LENGTH}$	+	[m]	2429.82	695.49	419.31	2407.19	789.22	454.11	-7.167	<0.001
$PATCH_{AREA}$	+	[ha]	183.77	18.36	20.35	183.77	20.90	21.92	-4.983	<0.001
Habitat parameters										
$FOREST_{DIST}$	-	[m]	1017.89	162.66	161.41	958.49	104.11	111.01	-13.307	<0.001
$AGRI_{AREA}$	-	[ha]	7.03	5.66	1.40	7.03	5.12	1.71	-11.082	<0.001
$URBAN_{DIST}$	+	[m]	2328.42	425.77	360.25	2472.24	563.93	368.26	-15.218	<0.001
$EDGE_{AREA}$	+	[ha]	6.77	2.27	1.83	6.99	2.90	1.64	-12.466	<0.001
ECO_{DIST}	-	[m]	1061.68	98.83	96.76	829.53	85.16	89.54	-5.640	<0.001
$FALLOW_{DIST}$	0	[m]	3883.35	899.89	697.88	4795.30	865.93	689.53	-1.848	0.065
$GRASS_{DIST}$	-	[m]	1061.68	135.96	127.83	1059.81	125.49	136.47	-5.077	<0.001
$ORCHARD_{DIST}$	-	[km]	9.90	2.26	2.06	10.03	1.97	1.92	-5.172	<0.001
$HEDGE_{DIST}$	+	[m]	1092.58	187.13	159.49	1288.76	204.64	166.90	-4.223	<0.001
$WATER_{LENGTH}$	0	[m]	793.91	83.95	135.14	790.50	77.22	133.11	-1.747	0.081
ASL_{RANGE}	+	[m]	138.8	33.18	22.52	120.6	38.21	23.37	-7.699	<0.001
ASL_{MEAN}	+	[m]	831.71	479.82	91.12	830.96	498.47	98.06	-7.150	<0.001
MSI	+	-	6.371	1.56	0.43	4.72	1.57	0.38	-3.083	0.002
$NEIGHBOUR_{DIST}$	-	[m]	1767.74	318.48	213.73	1621.65	225.23	221.52	-20.126	<0.001

Tab. 5.4: *Coefficients (B) of the binary logistic model fitted to the observations of brown hares and control points. Model coefficients are shown along with their standard errors and Wald-significance tests.*

	Coefficient (B)	SE	Wald- χ^2	df	P-value
$FOREST_{DIST}$	-0.00268	0.000	65.384	1	0.000
$URBAN_{DIST}$	0.00068	0.000	36.834	1	0.000
$NEIGHBOUR_{DIST}$	-0.00141	0.000	56.377	1	0.000
$RD12_{LENTGH}$	-0.00249	0.000	43.716	1	0.000
$RD456_{LENGTH}$	0.00038	0.000	16.868	1	0.000
$ORCHARD_{DIST}$	-0.00006	0.000	8.416	1	0.004
constant	*0.34755*	*0.135*	*6.677*	*1*	*0.010*

5.3.2 ROAD EFFECTS

DISTURBANCE EFFECTS

Correlation analysis between distance of plots to roads and hare densities per field area of each plot revealed a positive relationship, though this relationship failed to be significant. Mean density of hares was smallest in plots located in forest edge habitat adjacent to roads ($mean_{2005}$= 0.09 ha-1; $mean_{2003}$= 0.05 ha-1), and hare densities increased with increasing distance from the road up to 300m ($mean_{2005}$= 0.34 ha-1; $mean_{2003}$= 0.24 ha-1) (**Fig. 5.3**). In distances larger than 400m hare densities declined again ($mean_{2005}$= 0.22 ha-1; $mean_{2003}$= 0.11 ha-1). The pattern was the same for spotlight taxations in 2003 and 2005. However, following the results of ANOVA mean hare densities in road plots were equal among distance classes. This is because all distance classes showed large standard deviations, as hare densities were generally sparse in the study area and a large number of plots was not inhabited by hares.

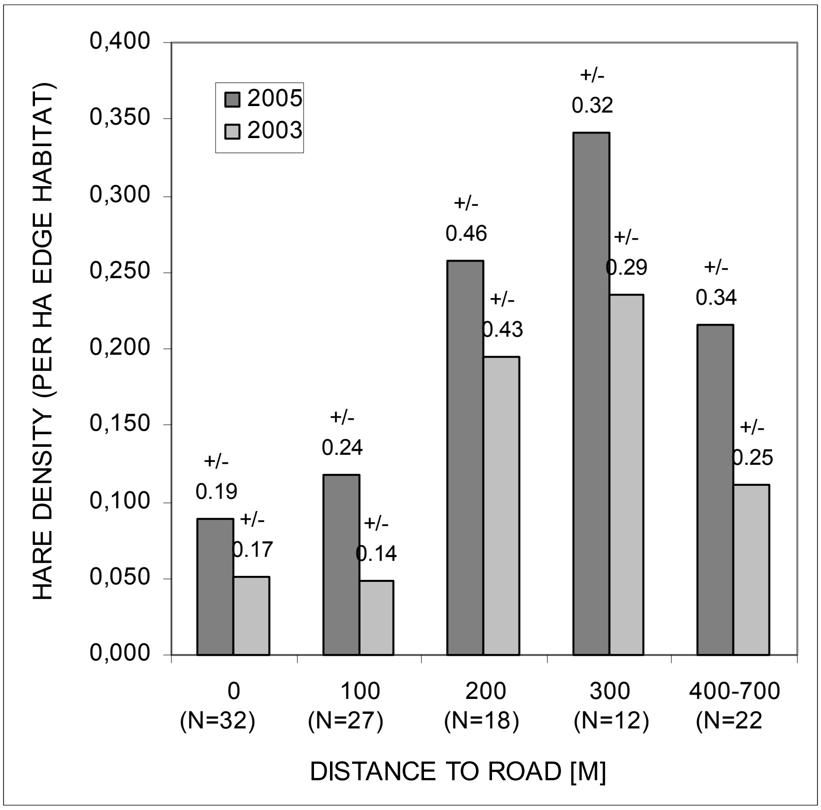

***Fig. 5.3**: Mean densities of brown hares in 111 longitudinal plots established in 5 distance classes parallel to roads. Road avoidance due to disturbance effects is observable up to 300m distance from the road. ANOVA testing equality of means among distance classes was not significant because of large standard deviations due to the generally sparse population abundance of brown hare in the study area.*

Barrier effects

Effects of roads on population abundance of hares, studied on raster grid level, have to be differentiated into road categories (**Fig. 5.4**). The density of highways, federal roads, main and side roads had a negative effect on abundances (R^2_{2005}= 0.42, p=0.012/ R^2_{2003}= 0.32, p=0.019). The density of drivable farm tracks and unpaved footways had a positive effect (R^2_{2005}= 0.43, p=0.011/ R^2_{2003}= 0.32, p=0.019). On the basis of the 4x4 km raster grid results were significant for hare abundances of 2005 and 2003. For the smaller 2x2 km raster grid variable values scattered and no significant relationship could be found.

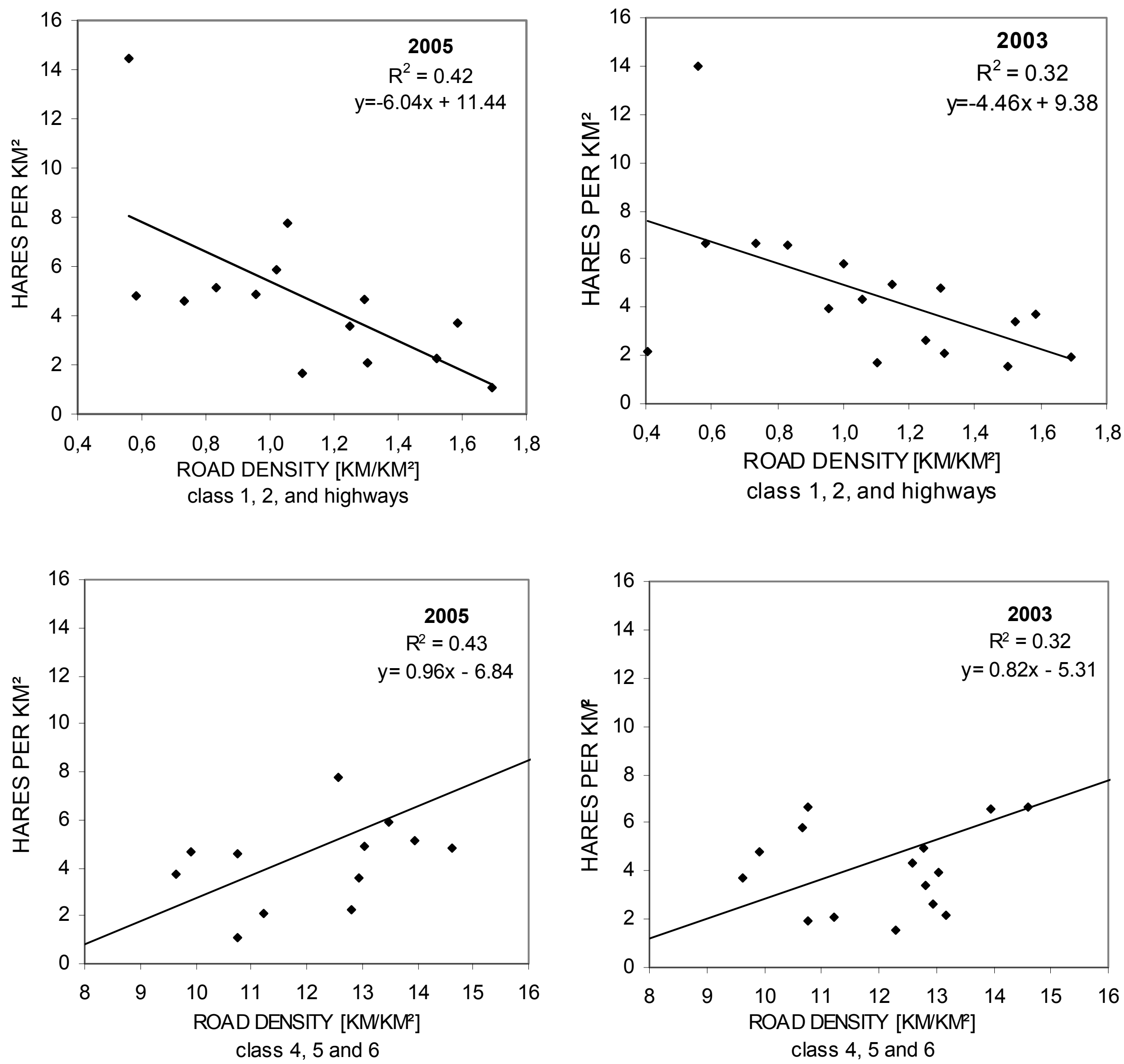

***Fig. 5.4**: Effects of road network density on abundance of brown hare in Canton Aargau. Base units for regression analysis are raster grid cells (4x4 km) with 20-30% forest edge area. Hare abundance is based on two spotlight taxations in 2005 (N=14) and 2003 (N=17). Road class 1 and 2: federal, main and side roads. Road class 4, 5 and 6: farm tracks and unpaved footways.*

ROAD MORTALITY

Hare populations in Canton Aargau showed three periods of historical declines, the first occurring between 1935 and 1940, the second between 1950 and 1955, and the third since 1980 (**Fig. 5.5**). With decreasing harvest and as a consequence of declining populations road mortality rates declined between 1981 and 1991 (**Fig. 5.6**). Road kills started to increase and outweigh harvest numbers since the 90ies. Between 2001 and 2003 road kills slightly decreased again. There was no significant relationship between road density of a hunting district and the amount of individuals killed by traffic, irrespective of road categories ($R^2<0.09$; $p>0.1$).

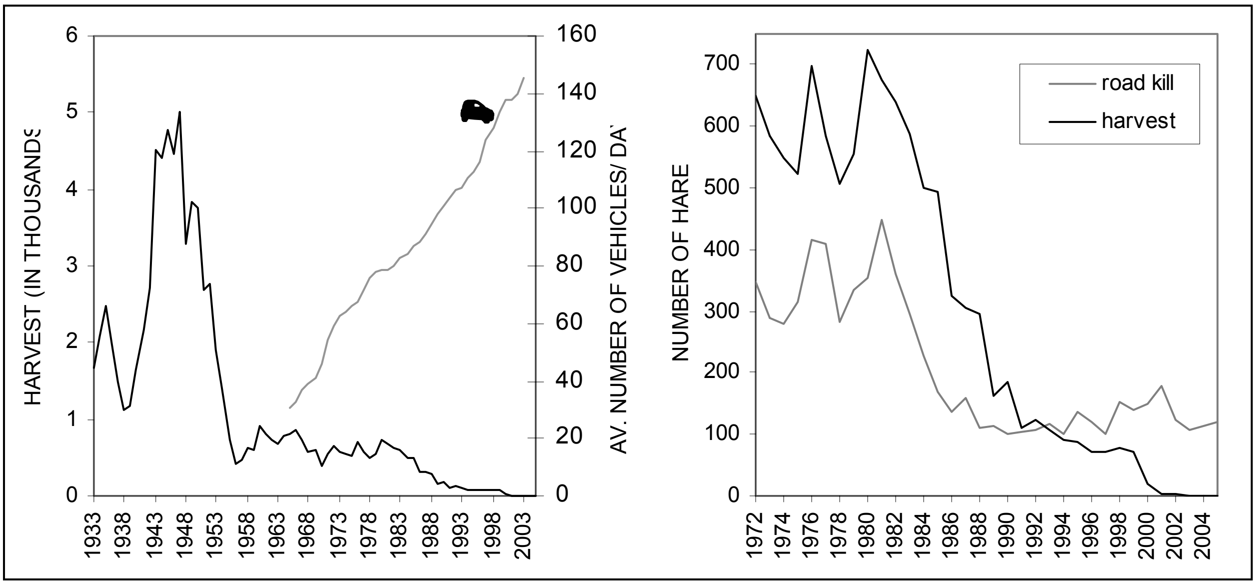

***Fig. 5.5 (left)**: Development of traffic volume (1965-2003) and brown hare harvest (1933-2005) in Canton Aargau in Switzerland, following the official hunting statistics (until 1952 incl. mountain hare (Lepus timidus)).*

***Fig. 5.6 (right)**: Development of brown hare harvest and road kills (1972-2005) in Canton Aargau in Switzerland, following the official hunting statistics.*

5.4 DISCUSSION

5.4.1 HABITAT PREFERENCES

We show that spatial distribution of brown hare is determined by a multiple impact factor complex, and these findings are supported by several other studies. Other authors showed that there is a significant relationship between spatial distribution and habitat diversity – highly variegated fields are more consistently inhabited by hares (Lewandowski and Nowarowski 1993, Tapper and Barnes 1986). Furthermore, hares prefer warm areas where soil dries up fast, and areas with cultivation of vegetables and grain (Pfister et al. 2002). In addition to this, we could show that hares frequent ecological buffer areas and semi-natural grasslands, while settlements are avoided (see also Bresinski 1983). According to our analysis, and following the results of Hoffmann (2003), hares also avoid hedges. However, the avoidance of hedges in our analysis seems to be a pseudo-correlation, as hedges are frequently planted next to roads in Canton Aargau.

The most important habitat factor for brown hare, according to our findings, is edge habitat (**Fig. 5.7**). Forest edges offer important seasonal and year-round rest, shelter and food areas when circumstances in fields are disadvantageous in an intensively used landscape (Rimathé 1977, Tapper and Barnes 1986). As it is crucial to control for habitat impact when analysing spatial distribution of brown hare with respect to roads, we exclusively analysed sites with a comparable amount of forest edge habitat in the continuative analysis. This approach is novel, as many other authors conducted multifactorial designs and were not able to make inferences about road effects.

Fig. 5.7: Cutout from the study area Canton Aargau in Switzerland showing two spotlight taxations of brown hare in 2003 (grey dots) and 2005 (black dots). Hares show a distinct tendency to prefer forest edge habitat as it offers important shelter and food.

5.4.2 Road effects

Disturbance effects

We show that hare density is smaller in plots close to roads than further away in the field. This result is a weak indication of disturbance effects arising for roads, although analysis of variance was not significant. Hare populations are too sparse in the study area to find explicit results in a plot study, however, disturbance effects have already been indicated by the habitat preference analysis showing that hares prefer large non-fragmented areas to smaller, more isolated patches, and avoid home ranges with a high density of highways, federal and main roads.

The avoidance patterns observed are not a methodological artefact caused by the taxation vehicle, because the spotlight taxation was conducted driving alongside field tracks, and not alongside the roads situated close to our study plots. Our results prove observations from Northern Germany, where smaller numbers of hares were seen in area adjacent to roads (150m) than in areas further away in the field (Strauß and Pohlmeyer 2001). Pfister et al. (2002) also showed that highly frequented roads with a high noise level are avoided. In our study, hare densities in distances larger than 400 m to roads decrease again. One reason for this may be that in larger distances the cumulative effects of the whole road network operate, and it is hard to detect the exclusive effect of a single road.

Barrier effects

Disturbance effects leading to the avoidance of the area next to roads may cause barrier and fragmentation effects at larger scales. It has been assumed in numerous places that habitat fragmentation affects population abundances of brown hare, for example, by hunters in Switzerland (Pfister et al. 2002). However, these large scale effects have not been proven yet. For example, in the study conducted by Pfister et al. (2002) hares were counted in 218 study areas throughout Switzerland. While showing that hare avoided the proximity of roads, no relationship was found between the values of an index for isolation and hare abundances at larger scales (Pfister et al. 2002). In another study in Switzerland key variables for hare densities were identified in 125 communes (Lundström-Gilliéron and Schlaepfer 2003). The authors provided evidence that the amount of vehicles affected hares, but area of roads and railways in the investigation area and lengths of national roads did not have any influence on hare abundance (regardless, the authors suggest using hare abundance as an indicator for the development of road and traffic networks). Furthermore, hares were counted in 30 study areas throughout Germany by spotlight taxations (Pegel 1986). Hunting intensity, predator density, climate, soil and several habitat attributes including the length of linear structures such as roads were analysed as potential impact factors, but no relationship between length of roads in study areas and hare abundances was found. A different study in Northern Germany did not find any relationship between density of road networks and population density of brown hare either, and the area with the densest road network was inhabited by the second largest population (Hoffmann 2003).

One reason for these findings may be that hare populations in response to disturbance alter their distribution, without any effect on population abundance. Likewise, it has been assumed that the reduction in bird densities near roads may be compensated by an increase in densities far away from roads (Illner 1992, Van der Zande 1980). The underlying hypothesis is that local scale disturbance effects may be compensated by landscape scale processes causing spatial shifts in abundance. However, we argue that a missing proof in former studies was mainly due to low quality of study designs. The strength of inference of these studies was limited with respect to roads for several reasons.

First, most of the studies prosecuted thus far, are multi-factorial population vulnerability analyses where study areas are compared showing a gradient of different habitat availabilities and environmental factors. When analysing population declines in study designs with multiple impact factors, secondary effects such as roads are suppressed by primary factors such as habitat availability. We used a study design excluding habitat availability as potential impact factor and were able to show that high densities of highways, main and secondary roads are correlated with low population abundances.

A second reason is the improper use of uninformative fragmentation indices. Instead of using road abundance as impact factor *per se*, indices with a high degree of complexity are calculated indicating several factors. The isolation index calculated by Pfister et al. (2002), for instance, is defined by field area size and constant isolation values assigned to different barrier elements. However, the same author found a positive correlation between field area and hare abundance, so the influence of fragmentation is superimposed by field area.

A third reason is that a majority of studies does not differentiate into road categories. This is a must indeed, as different road types ostensibly have different effects. We show on the one hand that the density of highly frequented roads has a negative effect on hare abundance, and this may be a consequence of the road avoidance behaviour observed at local scales in the plot study. In contrast, areas with a high density of field tracks and unpaved footways are preferred. The vegetation at field edges seems to be an important food supply during periods of hunger caused by harvest (Lewandowski and Nowarowski 1993, Schröpfer and Nyenhuis 1982). At first sight the results concerning field tracks seem to be contradictory. Although hares prefer areas with a high density of tracks, a comparison of hare and control locations in a 150m buffer showed that the proximity to farm tracks and unpaved footways ($ROAD456_{DIST}$) seems to be avoided as well. However, we think that this is an artefact of measurement accuracy, because the difference between hare and control points is too small (3m) to make inferences. It was observed in the field that hares stopped social activities and feeding, and took flight when a vehicle approached following a fixed route on field tracks. We assume that the avoidance of field tracks observed in our habitat preference study was mainly caused by presence of taxation vehicles, and that the vegetation at field tracks is frequented again when the vehicle drives on.

MORTALITY EFFECTS

Hares move between habitats daily and shift activity between fields according to crop development (Tapper and Barnes 1986). When moving in-between fields in search of food or mates requires crossing a road, they are vulnerable to road mortality due to collisions with cars. However, we did not find any relationship between the density of roads in a hunting district and road kill rates. However, interpretation of road kill data is problematic as road mortality is a function of population abundance (Eskens et al. 1999, Strauß and Pohlmeyer 2001). On the one hand high mortality rates may decimate abundances, and on the other hand high abundances will probably lead to high road mortality rates (the more animals, the more potential road crossings). This problem is exemplified by harvest statistics used in our analysis. Decreases in harvest after 1980 do not reflect population declines, but rather the abdication of hunting as a reaction to declines. Hence, increases in road kills after 1985 may reflect a slight increase in populations exempt from hunting pressure. However, increasing road kills may also be a consequence of increasing traffic densities, a trend which has been shown for Switzerland and Germany (Pegel 1986, Pfister 1990). We conclude, that the use of hunting statistics in our analysis does not allow inferences about the effects of road networks on mortality, and the consequences on population abundances thereof.

5.5 CONCLUSIONS AND IMPLICATIONS

Hare populations in the study area have been in decline three times during the last century, likewise in many other areas in Europe during the last decades. Causes for these declines have been analysed in numerous studies, and relationships between different factors have been quantified in various ways at different spatial scales (Smith et al. 2005). Although conflicting results have been obtained, there is compliance about a multi-factor complex including weather conditions, diseases, predation, hunting and habitat quality (Strauß and Pohlmeyer 2001). High abundances can probably be expected in mild winters, in areas with low precipitation and low predation, and in agricultural fields of high diversity (Smith et al. 2005). Cold summers with high precipitation, in particular, lead to high losses in offspring caused by bacterial and parasitic infections such as pseudotuberculosis and coccidiosis (Spittler 1987, Strauß and Pohlmeyer 2001). As a consequence, such extreme weather conditions can lead to short-term strong reductions in abundance. These temporal declines can be compensated under ideal circumstances. However, in unfavourable situations they cause irreversible losses and may lead to the extinction of a local population. As soon as the local population is debilitated by extreme weather conditions, roads become an important impact factor. As hare populations are currently in decline and susceptible to various impact factors, understanding the role of potential cumulative effects, such as roads, becomes more and more important.

We excluded habitat as potential impact factor and analysed the exclusive effects of roads on hare populations. We were not able to show any effect of road density on road mortality. However, our study demonstrated disturbance effects arising from roads. At local scales these lead to the avoidance of the area adjacent to roads, and at large scales to decreases in population abundances. Against this background it is questionable if current measures constructed to mitigate road effects are effective, as they mainly focus on mitigating road mortality. Further research will have to figure out whether disturbance effects are caused by the physical presence of vehicles, noise, light, or the indirect loss of habitat due to alterations in the physical and chemical environment adjacent to roads. Based on such studies new mitigation strategies will have to be developed addressing disturbance effects. Furthermore, our results indicate that hares prefer natural habitats such as ecological buffer areas and fallow land. Consequently, road effects on population abundance might be compensated by advancements in habitat quality.

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Chapter 6.
Temporal and spatial characteristics of vehicle-wildlife accidents in Germany

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Abstract

Vehicle-wildlife collisions are a serious problem because of animal welfare, economic and traffic safety reasons. We aimed at predicting time and location of such accidents to increase the effectiveness of mitigation strategies. We analysed a database of 1,726 records of vehicle-wildlife accidents in Hesse (Germany) concerning spatio-temporal characteristics. First, we used the network K-function method to describe the spatial pattern of accident locations on the road network. Second, we measured 27 landscape, population and road parameters within a 0.5km radius of each site on the basis of digital road and landscape data to determine landscape characteristics at accident locations. Third, we developed four models by means of logistic regression analysis comparing accident sites with randomly selected control sites. Temporally, accidents most frequently occurred after sunset and at sunrise in early morning hours. Seasonal peaks occurred in May, July-August and October. Spatially, accident sites were in close proximity to forest coverage. Hotspots for collisions were woodland-field interfaces frequented by animals when changing between forest habitat and forage areas on open lands. With respect to road characteristics accidents occur at intermediate traffic densities. One model including all parameters distinguishing significantly between accident and control sites correctly classified 72.5% of 252 sites not included in model building. Predictive models should be used for an assessment of possible high-risk locations and for planning large-scale, statewide migration corridors in the context of defragmentation programs. Results are of high practical relevance as this is the first study analysing collision sites in a Central European landscape.

Zusammenfassung

Raum-zeitliche Charaktersitika von Wildunfällen in Deutschland. *Kollisionen zwischen Fahrzeugen undWildtieren sind ein ernstzunehmendes Problem aus Sicht von Tierschutz, Ökonomie und Verkehrssicherheit. Die vorliegende Untersuchung zielt darauf ab, Zeit und Ort von Wildunfällen vorauszusagen, um die Effektivität von Vermeidungsstrategien zu erhöhen. Verwendet wird ein Datensatz von 1726 polizeilich gemeldeten Wildunfälle in Hessen, die hinsichtlich raum-zeitlicher Besonderheiten untersucht werden. Zunächst wird die netzwerk-basierte K-Funktion verwendet, um räumliche Muster von Unfällen auf dem Straßennetz zu beschreiben. Zur Ermittlung raum-zeitlicher Charaktersitika werden 27 Straßen- und Landschaft parameter in einem Radius von 500m um Unfallorte erhoben. Es werden vier Modelle mittels logistischer Regression entwickelt, die Unfallpunkte mit zufällig verteilten Kontrollpunkten vergleichen. Zeitlich gesehen finden Wildunfälle am häufigsten nach Sonnenuntergang und vor Sonnenaufgang in den frühen Morgen-stunden statt. Jahreszeitlich treten Schwerpunkte im Mai, Juli-August und Oktober auf. Räumlich gesehen befinden sich Unfallorte in der Nähe zum Waldrand. Schwerpunkte sind Wald-Offenland Übergänge, die Tiere auf dem Weg zu Äsungsplätzen im Offenland frequentieren. In Bezug auf Straßencharakteristika sind Wildunfälle häufig bei mittlerem Verkehrs-aufkommen anzutreffen. Ein Modell, das alle signifikant zwischen Unfall- und Kontrollpunkt unterscheidenden Parameter enthält, klassifiziert 72.5% von 252 Validierungspunkten richtig. Die beschriebenen Vorhersagemodelle sollten zur Lokalisierung von Unfallschwerpunkten genutzt werden, insbesondere zur großflächigen Planung von Wildtierkorridoren im Rahmen landesweiter Ent-schneidungsprogramme. Die Ergebnisse sind von hoher praktischer Relevanz, da die vorliegende Studie für mitteleuropäische Landschaften bislang einzigartig ist.*

6.1 INTRODUCTION

Traffic mortality is one of the most obvious negative impacts roads have on wildlife species. Numbers of road killed animals are exceeding worldwide: The US nationwide road-kill for 1991 totalled at least 500,000 - 726,000 animals, potentially exceeding 1 million if including unregistered accidents (Conover 1995; Romin and Bissonette 1996). In Europe, approximately 500,000 collisions occur each year (Groot-Bruinderink and Hazebroek 1996), and in Germany the nationwide road-kill for 2005/06 totalled 199,900 roe deer, 22,334 wild boar and 2,714 red deer individuals (Deutscher Jadgschutz Verband 2006).

For wildlife, most accidents are fatal, e.g., 92% of roe deer involved in accidents were killed in Michigan (Allen and McCullough 1976). Though mortality rates are high in Germany, they are sustainable for roe deer and wild boar populations, as mortality attributable to road-kills does not exceed the replacement rate from reproduction or immigration. However, collisions between vehicles and large wildlife species such as roe deer and wild boar are a serious problem, because of animal welfare purposes, economic and traffic safety reasons. Human injury results from approximately 4% of collisions, involving medium-sized animals such as deer (Conover et al. 1995) and from 14% to 18% of collisions with larger mammals like moose (Joyce and Mahoney 2001). In Germany, 2,291 accidents were followed by personal damage to vehicle occupants in the year 2005, 578 people were badly injured, and 14 accidents were fatal (Statistisches Bundesamt 2005). The mean material damage for cars caused by wildlife accidents has been estimated to be 1,903 Euro in Germany, resulting in 425 million Euros reported to comprehensive insurances (Deutscher Jagdschutz Verband 2004), and totalling over 1 billion Euros in Europe (Groot-Bruinderink and Hazebroek 1996).

The trend in Germany is increasing as a consequence of constantly growing roe deer and wild boar populations, and the ongoing densification of road networks (Roedenbeck and Köhler 2006). To counterbalance this development, predicting time and location of vehicle wildlife collisions would be helpful. The knowledge of temporal and spatial characteristics at accident locations is crucial to recommend locations for mitigation efforts, because if mitigation measures, such as fences, road signs and passageways are constructed at the wrong place they fail to be effective (Reeve and Anderson 1993; Putman 1997).

There is accumulating scientific evidence that wildlife accidents do not occur randomly in time and space. It has been assumed that accident times are associated with behavioural patterns of wildlife and accident locations are related to habitat requirements. When trying to apply these results in Germany, three characteristics of relevant literature stand out: (i) Most of the published studies have been carried out in the USA (Bashore et al. 1985; Finder et al. 1999; Hubbard et al. 2000; Nielsen et al. 2003), a few in Sweden (Seiler 2005) and Spain (Malo et al. 2004) despite the urgency of the problem in Europe. Hence, there is no study including Central European landscape characteristics. (ii) Parameters influencing wildlife accidents seem to differ regionally, depending on the species and landscape under investigation, constraining extrapolation from previous studies. (iii) All studies use differing methods and scales, for example, they investigate a wide range of different landscape variables (**Tab 6.1**). Some studies are based on GIS landscape data, others on field surveys (**Tab. 6.2**). Some studies investigate road characteristics in the direct proximity to accident locations, others use a buffer based analysis, and others investigate collision hotspots based on kernel density estimates (**Tab. 6.2**). Consequently, comparability among studies is hardly achievable.

The specific objective of this study was to investigate vehicle wildlife accidents in a Central European landscape (Germany) with regard to temporal and spatial characteristics. We developed models predicting time and location of such accidents. Predictive models are the basis for addressing mitigation hotspots in a statewide defragmentation programme. A special task of our analysis was to base models on digital landscape data being readily available, so that model application is possible in road planning offices. A second objective was to compare our results with the studies from the USA, Sweden and Spain in order to name Central European peculiarities (**Tab 6.1** and **6.2**).

6.2 MATERIAL AND METHODS

6.2.1 STUDY AREA

The study was carried out in the federal state of Hesse, Germany. Topography and landscape relief of the 21,100 km² state is hilly and consists of several low mountain ranges embracing plains, lowlands and plain tracts. Altitude ranges from 80 to 950 m above see level (ASL). The climate is warm temperate with mean precipitation between 400 and 1,000 mm. Land use mainly depends on altitude, with intensive agriculture (43%) and urban areas (15%) in plains, and forestry (40%) in peripheral low mountain ranges. Forests mainly consist of mixed forest (56%), followed by deciduous forest (26%) and coniferous forest (17%). The network of public roads covers 16,733 km (of which 962 km are motorways), resulting in a density of 0.79 km public road per km² land area. Vehicle density in 2006 was 592 per 1,000 residents. Average daily traffic varies between 400 vehicles in peripheral regions and up to 150,000 vehicles on motorways. Landscape fragmentation by roads has increased in Hesse since 1930 as well as population densities of roe deer (*Capreolus capreolus*) and wild boar (*Sus scrofa*) (Roedenbeck and Köhler 2006). Approximately five people are killed per year by accidents with wildlife in Hesse, 400 get injured, and the property damage totals at least 45 million Euros per year (Deutscher Jagdschutz Verband 2006).

***Tab. 6.1**: Parameters investigated in studies predicting locations of vehicle-wildlife accidents. The relationship between variable values and collision probability may be: (+): positive, (-): negative, (s): significant (without indicating + or -) or (0): not significant.*

	Roedenbeck & Köhler (2007)	Malo et al. (2004)	Madsen et al. (2002)	Nielsen et al. (2003)	Hubbard et al. (2000)	Finder et al. (1999)	Bashore et al. (1985)	Puglisi et al. (1974)	Bellis & Graves (1971)	Seiler (2005)	Joyce & Mahoney (2001)	Clevenger et al. (2002)	Ramp et al. (2005)
Species	**roe deer, wild boar**			**White-tailed deer**						**moose**		**others**	
Road and traffic													
traffic volume	x (0)		x (0)		x (0)					x (s)	x (+)	x (0)	
speed limit		x (0)		x (0)			x (-)			x (s)	x (+)		
curvature	x (-)	x (0)		x (0)		x (0)					x (0)		x (s)
visibility		x (0)					x (+)						
type of road	x (s)											x (s)	
roadside shape (banks, gullies, level)		x (s)				x (s)	x (0)	x				x (s)	
number of lanes				x (0)	x (+)								
number of barriers in roadway												x(0)	
guard rails		x (-)					x (0)		x (0)				
cross roads		x (-)								x (+)			
topography, elevation, slope	x (+)	x (0)		x (0)		x (+)	x (0)	x (s)	x (0)	x (0)		x (-)	x
density of roads (in buffer)	x (-)									x (+)			
density of railroads (in buffer)	x (-)									x (-)			
Road median and verge													
median/ verge (area, abs/pres, quality/type of vegetation)		x (0)							x (0)			x (0)	
density of vegetation			x (+)										
Settlements													
Developed/urban land (percentage or area)	x (-)	x (-)		x (-)	x (0)					x (-)			
residences and buildings (number or density)				x (0)		x (+)	x (-)			x (0)			
distance to town or house	x (+)	x (+)			x (0)	x (0)						x (-)	x
Habitat amount (area, density, patch size or proportion of transect lines)													
forest	x (+)	x (+)		x (+)	x (+)	x(+) *	x (0)	x		x (+)		x (0)	x
forest-open mixed												x (0)	
non-wooded, shrubs		x (0)					x (+)						
forage (crops, fields, orchards)	x (0)	x (-)			x (s)	x (-)		x		x (-)			
open (grassland, pasture, bare soil)	x (0)	x (0)		x (0)	x (+)	x (+)	x (0)	x		x (+)		x (0)	x
fellow		x (0)						x					
public land				x (+)									
wetland										x (0)			
recreational land						x(+) *							
Habitat structure and quality													
distance to forest or tree lines	x (-)	x (-)		x (0)		x (-)	x (-)	x		x (-)		x (-)	x
distance to hedges or number of hedgerows		x (-)				x (0)							
distance to recreational land (park, refugial)	x (-)					x (-)							
field/forest edges (number or density)	x (+)					x (+)							
similarity of road sides	x (s)							x (s)					
intersection of road with forest edge, water courses										x (0)			
ecotone length		x (0)											
attraction points (pond, feeding points)		x (0)											
non-fragmented area on both road sides (area size)	x (-)												
corridors (distance to)	x (-)												
corridors (width)						x (+)							
number of bridges					x (+)								
habitat diversity (Shannon´s or Simpson´s diversity index)		x (+)		x (+)		x (+)				x (+)			
landscape indices: edge, shape, core area, interspersion etc.	x (s)			x (0)	x (s)	x (s)				x (s)			
Water availability													
water (area or presence)	x (0)			x (0)	x (0)	x (+)						x (0)	
distance to water	x (0)	x (0)										x (0)	x
water availability index	x (s)												
river density	x (0)	x (0)								x (0)			
Mitigation measures													
fences		x (0)					x (-)	x (s)	x (0)	x (-)			
crossing structure (pres/abs or distance)		x (-)										x (+)	
density of passages across road										x (0)			
distance to warning sign		x (0)											
Populatzion size													
Population size	x (+)							x (+)	x (0)	x (+)	x (s)		
Further information													
time	3 peaks: May, Jul-Aug, Oct		3 peaks: Feb-Mar, Jun-Jul, Oct		2 peaks: May, Nov			2 peaks: May-Jun, Nov-Dec	2 peaks: Apr-May, Nov-Dec		1 peak: Jul-Aug	1 peak: Apr (mammals)	
time	peak at dusk, night, dawn										peak at dusk, night, dawn		

***Tab.6.2**: Comparison of studies investigating spatial characteristics of vehicle-wildlife accidents.*

	Roedenbeck & Köhler (2007)	Malo et al. (2004)	Madsen et al. (2002)	Nielsen et al. (2003)	Hubbard et al. (2000)	Finder et al. (1999)	Bashore et al. (1985)	Puglisi et al. (1974)	Bellis & Graves (1971)	Seiler (2005)	Joyce & Mahoney (2001)	Clevenger et al. (2002)	Ramp et al. (2005)
Species	roe deer, wild boar	roe deer, red deer, wild boar	roe deer (*Capreolus capreolus*)	deer (*Odocoileus spp.*)	white tailed-deer (*Odoc. virinianus*)	white tailed-deer (*Odoc. virinianus*)	white tailed-deer (*Odoc. virinianus*)	white tailed-deer (*Odoc. virinianus*)	white tailed-deer (*Odoc. virinianus*)	moose (*Alces alces*)	moose (*Alces alces*)	small terrestrial vertebrates	kangaroos, wallabies, wombats, birds
Study area	Hesse	Soria Province	East Jutland	Minnesota	Iowa	Illinois	Pennsylvania	Pennsylvania	Pennsylvania	South-central	Newfoundland	Alberta	New South Wales
	Germany	Spain	Denmark	USA	USA	USA	USA	USA	USA	Sweden	Canada	Canada	Australia
Vegetation	forest, urban land, agricultural fields	forest, dry land herbaceous crops, long-term abandoned crops, grasslands	forest, agricultural fields separated by hedgerows	residential and commercial properties, greenspace areas, oak forest	mainly agriculture, small fields interspersed with pastures and mixed hardwood forest	no specification	narrow mountain ridges with mixed hardwood forests and pine, residents in valleys	oak-hickory forest, farms, agricultural fields	decidous forest, mixed hardwood, no residents	dominated by boreo-nemoral conifer-dominated forest, few agricultural land use	boreal forest, hog-marshland, rocky highlands, abundant lakes and rivers	montane - subalpine, open forest, natural grasslands	mountainous, cleared land, moist and dry forest
Study period	1995-2004	1988-2001	1967-87	1993-2000	1990-97	1989-93	1979-80	1970-72	1968-69	1972-99	1988-94	1997-02	1998-2004
N (kill sites)	1726	41 high collision sites, 220 collision points	115	80	466	86	51	870	286	2000	1690 accidents, 90 road segments	604 (272 mammals)	ca. 1000
method	GIS-based analysis, 500m buffer	field data and GIS-based analysis at 2 spatial scales: 1) high collision sections with 1000m buffer, 2) collision points with 100m buffer	just few spatial variables derived by GIS: distribution of accidents in space (no buffer)	GIS-based analysis, 100m buffer	GIS-based analysis, 800m buffer	GIS-based analysis, 800m buffer	field data, mapping with the help of transects perpendicular to roads (no buffer)	analysis of reports (field data of game protectors)	field data (no buffer)	GIS-based analysis, 500m buffer	just few spatial variables derived by GIS: moose-density and traffic volume (no buffer)	field data and maps (no buffer)	GIS-based analysis, buffer according to species´ homerange size
kill site defintion	GKK of kill sites	1) midpoint of 1km road segments with >3 accidents per km, 2) each collision point	150m road sections, number of accidents per segment	500m road segments, >=2 accidents per segment	1,61km milepost markers, randomly selected out of 9575 mileposts, numbers of accidents per milepost, >14 accidents per 7 years	1.3km road segments, numbers of accidents per segment, >15 accidents per 5 years	accident sites of varying length, >2 accidents per year	numbers of deer killed per mile	200 feet road segments, number of accidents per road segment	GKK of kill sites	10km road segments, number of accidents per segment per year, 3 classes: <1,75 low, 1.75 - 3.0 medium, >3 high	accidents as GKK, calculation of a road kill index, frequency per 1000km	kernel polygon of all accidents sites per species
control site definition	random points, >1km distance to kill site	1) midpoint between the end of two high collision sections, 2) random points	no control sites	random road segments, 0-1 accidents	segments with 0-13 accidents per 7 years	paired to kill sites (on same street) to minimize variation in traffic volume and deer density, 800-8000m distance from kill site	paired to kill sites (on same street), 100-1200m distance from kill site, less than 20% collisions than kill site	no control sites	no control sites (numbers of deer killed are correlated with parameters)	random points, >1km distance to kill site	no control sites	random points	random points
N (validation sites)	252 (global model: 69.1%; road model: 59.9% correctly predicted)	1) 30 (87% correctly predicted), 2) 95 (74% correctly predicted)	-	40 (77.5% correctly predicted)	245 (63% correctly predicted)	10 (model 1: 90%; model 2: 60% correctly predicted)	10 (90% correctly predicted)	-	-	2600 (combined model: 76.1%; traffic model: 77.9% correctly predicted)	-	-	-

6.2.2 Data

We used an official database on vehicle collisions with wildlife maintained by the Hessian State Office for Roads and Traffic for the period between 1995 and 2004. The statistics include police-reported vehicle-wildlife accidents with personal injury of vehicle occupants or severe property damage. From a total of 2,111 records 1,726 include accident date and time, location as spatially explicit Gauss-Krueger coordinate, road category in four classes from county road up to highway, and speed limit if given at accident location. We can infer that collisions were mainly caused by roe deer (*Capreolus capreolus*) and wild boar (*Sus scrofa*), since only accidents with severe damage are included in the data.

Road and landscape data were obtained from ATKIS-Basis-DLM (DLM2/2002), a digital GIS-database derived on the basis of topographical maps and digital orthophotos. The data distinguishes between major land cover types and includes linear elements such as roads, railways and rivers. Spatial accuracy of linear elements is +/-3 m, and for planar land cover elements +/- 0.5-5 m. Traffic density on roads was obtained from a traffic density map digitized by the Hessian State Office for Roads and Traffic (HLSV 2002). Landscape topography was obtained from a digital elevation database (DGM25) with a spatial resolution of 40 x 40 m cells.

6.2.3 Spatial clustering of accidents

To describe the distribution of accident locations on the road network we applied a kernel density estimate to evaluate where collisions were mostly occurring. Furthermore, we used the network K-function method, which is an adequate adaption of Ripley´s K-function (Ripley 1977; Okabe and Yamada 2001). While Clevenger et al. (2002) and Ramp et al. (2005) both used the K-function to analyse clustering of wildlife accidents on a single road, we introduce the method to investigate spatial distribution of wildlife accidents on a road network (Okabe and Satoh 2006). The network K-function can be interpreted as the total number of points located within a given shortest network distance d of each point, corrected for the overall point density of the network. The process is based on the hypothesis that points are independently distributed over the network. Thus, if this hypothesis is rejected, points are spatially interacting and may form uniform patterns.

The observed function $K_{obs}(d)$ was calculated for increasing values of d, and compared to the expected function $K_{exp}(d)$ showing values that would be expected if the accidents were randomly distributed along the network. Both functions were calculated using SANET 3.0 extension for ArcGIS 9.0 (Okabe et al. 2006). The analysis was conducted for all highways in Hesse, and could not be conducted for the whole network, because size limitations exist for the software. We specified d as an interval of 50m and used 100 Monte Carlo Simulations to construct a confidence envelope based on the maximum and minimum values from an equivalent number of random points for $K_{exp}(d)$. If values of $K_{obs}(d)$ lie within the confidence envelope, points are assumed to be randomly distributed. If values of $K_{obs}(d)$ lie above the upper, or below the lower confidence interval, points tend to cluster, or towards regularity, respectively (Bailey and Gatrell 1995; Spooner et al. 2004; Deckers et al. 2005).

6.2.4 VARIABLES ASSOCIATED WITH ACCIDENT SITES

Landscape, traffic and population characteristics associated with accidents were analysed on the whole road network of Hesse using Geographical Informationing Systems (GIS) ArcView 3.2 and ArcGIS 9.0 (ESRI). We compared 1,726 accident sites with 1,726 randomly selected control sites. Control sites were randomly distributed on the road network by the GIS-Tool Random Point Generator (Jenness Enterprises 2005). The random selection of absence points was not constrained by where presence locations were, however, we set a minimum separation distance of 1,000m between accident and control sites to avoid an overlap of landscape characteristics.

We then established a buffer of 500m radius around each control and accident site to account for some presumed error in the reported location (Seiler 2005), and calculated 27 different road, landscape and population parameters potentially influencing the risk of collisions (**Tab. 6.3**).

First we calculated several land use variables, such as proportion of urban area, proportion of forest, deciduous forest, conifer forest, agriculture and open land, referring to buffer area size. In addition, we measured the distance from each point to the nearest forest edge and settlement. We used *Patch Analyst Extension* for ArcView 3.2 to compute several landscape indices for forest habitat (McGarigal and Marks 1995; Elkie et al. 1999), such as *mean patch size* and *mean shape index* of forest patches as well as *edge density* of forest edges in each buffer.

To describe the similarity of road sides we divided each buffer into two halves separated by the accident road. We then estimated the proportion of forest on both buffers sides and measured similarity as the difference in forest proportion between both sides as absolute value. Hence, values of similarity index range between 0 and 100 with 0 indicating absolute similarity, i.e. same proportion of forest on both sides, and increasing values indicate decreasing similarity (**Tab. 6.3**).

To describe habitat quality we measured the distance of each site to the next nature conservation area and landscape corridor. We included all nature conservation areas protected by the flora-fauna-habitat (FFH) directive (Council of the European Union 1992). Location of corridors was obtained from a GIS-Shapefile provided by the German Hunting Association (DJV) indicating potential migration corridors for large forest species (see Schadt et al. 2002; Reck et al. 2004).

Characterising the availability of water, we estimated proportion of lakes and length of water courses per buffer area, distance to next water course and an index describing the availability of water on both road sites.

As an index for population size of roe deer and wild boar we used average annual game bag per county in 2002/03, giving 21 density classes.

As road and traffic characteristics, we calculated traffic density as average number of vehicles per day, length of roads and railways in buffers, and road category in four classes from county road up to highway. As an index for visibility on roads we measured variation in topography as the maximum altitudinal difference in each buffer. We also calculated an index of road curvature describing the length of the accident road in the buffer divided by 1000. An index of 1 means no curvature and increasing values indicate increasing curvature.

To describe landscape fragmentation we calculated the area size animals can use on both road sides without crossing another road or entering a settlement.

Tab. 6.3: *Environmental parameters measured at vehicle-wildlife collision sites and control sites in Hesse (1995-2004). Variables are grouped into 7 factors by principal component analysis (PCA). We show the percentage of variance explained by each factor.*

Factors and variables	measure	definition
Factor ´land use´ (19.1 %)		
$FOREST_{PROP}$	[%]	Proportion of forest in 500m buffer
$OPEN_{PROP}$	[%]	Proportion of open land (agriculture and grassland) and in 500m buffer
$DECID_{PROP}$	[%]	Proportion of deciduous forest in 500m buffer
$CONIFER_{PROP}$	[%]	Proportion of coniferous forest in 500m buffer
$AGRIC_{PROP}$	[%]	Proportion of agricultural fields in 500m buffer
$FOREST_{DIST}$	[m]	Distance to nearest forest edge (0 if in forest)
$FOREST_{ED}$	[m/m²]	edge density of forest edges in 500m buffer
MSI	-	mean shape index of forest patches in 500m buffer, index >1
MPS	[ha]	mean patch size of forest patches in 500m buffer
Factor ´road´ (13.1 %)		
$ROAD_{KM}$	[km]	Length of roads (country roads up to highways) in 500m buffer
TRAFFIC	[cars/d]	Traffic density measured in thousand cars per day
$ROAD_{CAT}$	-	Road category in 4 classes from county road up to highway; transformed to a binary variable for regression analysis (1: highway; 0: no highway).
Factor ´urban´ (8.7 %)		
$URBAN_{PROP}$	[%]	Proportion of urban areas in 500m buffer
$URBAN_{DIST}$	[m]	Distance to nearest town periphery (0 if in town)
UNFRAG	[km²]	Size of non-fragmented areas adjacent to both roadsides (0 if in town)
$TRAIN_{M}$	[m]	Length of railways in 500m buffer
Factor ´population´ (5.6 %)		
CAPREOLUS	[#/Km²]	Roe-deer abundance, number of individuals hunted per km² per year (0.32-3.57).
SUSSCROFA	[#/Km²]	Wild boar abundance, number of individuals hunted per km² per year (0.03-2.75).
$CORR_{DIST}$	[km]	Distance to nearest wildlife corridor (corridors for large forest species)
FFH_{DIST}	[km]	Distance to nearest protected area (Flora-Fauna-Habitat FFH areas – 0 if inside)
Factor ´water´ (5.0 %)		
$LAKE_{PROP}$	[%]	Proportion of lakes in 500m buffer
$WATER_{KM}$	[km]	Length of water courses in 500m buffer
$WATER_{DIST}$	[m]	Distance to nearest watercourse or lake
$WATER_{AVAIL}$	-	Index for availability of water (1: lake on both sides; 0: lake on no, or on one side, respectively).
Factor ´visibility´ (4.0 %)		
$CURVE_{ID}$	-	Road curvature: length of accident road in buffer/ 1000, 1: no curv., >1 curv.
ASL_{RANGE}	[m]	Variation in topography, distance between highest and lowest elevation in buffer
Factor ´similarity´ (3.7 %)		
$SIMILAR_{ID}$	-	Index for similarity of road sides concerning the amount of forest SIMILAR ID = |Δ FORESTPROP | , 0 ≤ SIMILAR ID ≤ 100 (0: similar; 100: not similar)

6.2.5 PREDICTIVE MODELS

We generated predictive models for the location of accident and control sites using binominal logistic regression analysis, with site status as binary response variable (accident: 1, control: 0). 1,600 accident and 1,600 control sites were randomly selected for model building. The remaining 126 accident and 126 control points were later used for model validation.

To find reasonable variable combinations for model building we conducted two different variable reduction procedures. First, we checked the differences between accident and control sites (N=3,452) by comparing the estimates of metric variables with non-parametric Mann-Whitney-U-Test and categorical variables with χ^2- test. As the sample size was large we calculated significance as well as *Cohen´s d*, as a measure of effect size. We interpreted effect sizes in categories of small (d<0.2), medium (d=0.2-0.5), and large (d=0.5-0.8) (Valentine and Cooper 2003). Second, we used principal component analysis (PCA) to investigate variable interaction, and grouped all variables into 7 ´factors´ (**Tab. 6.3**).

Following the results of the variable reduction procedures we grouped all variables into four a priori models. (i) The GLOBAL model included all variables selected by U- and χ^2- tests to obtain a preliminary equation. (ii) In the FACTOR model we selected representative variables of each factor derived by PCA (**Tab. 6.3**), two variables from each of the first four factors and one variable from each of the last three factors. This procedure reduces the number of model parameters by exclusion of interacting variables. (iii) The ROAD model included, next to the proportion of forest and urban area, all parameters describing road and traffic characteristics. This model tested, if accident sites are predictable by only using road related parameters which are readily available in road offices. (iv) The HABITAT model was based on hypothesized importance to deer and wild boar ecology. Though other variable combinations would have been possible we restricted further analysis to these four models, as they are based on appropriate variable reduction techniques, data availability and animal ecology.

We ranked and compared all four models using *Akaike´s Information Criterion* (*AIC*) and the associated statistics *delta AIC* (Δ_i) and *Akaike weights* (w_i) (Anderson et al. 2001; Akaike 1974; Burnham and Anderson 2002). *AIC* penalizes for the addition of parameters into the model, and thus selects a model that fits well but has a minimum number of parameters. The model with lowest *AIC* was chosen as "best" model according to simplicity and parsimony.

6.2.6 MODEL VALIDATION

Models were validated with the 252 accident and control points not included in the model building process. Values of these points were entered into the model equations to see whether each location was an accident or control point. To select a threshold by which validation sites were classified, we used *receiver-operating-characteristics* (ROC curves) (McNeil and Hanley 1984). The area under the ROC curve is a visual index of model accuracy, and the further the curve lies above the reference line, the more accurate the model. The cutoff value for classification was set where the number of correct classifications was maximised. For this cutoff value we give the sensitivity (number of correctly predicted accident sites/ number of all accident sites), the specificity (number of correctly predicted control sites/ number of all control sites), and the overall concordance of correct classifications. Statistical analyses were performed using SPSS 11.0.1 (LEAD Technologies Inc., Chicago) and SAS 8.2 (SAS Institute Inc. 1999).

6.3 RESULTS

6.3.1 TEMPORAL PATTERNS

In total, 2,111 wildlife accidents with severe personal or material damage were recorded between 1995 and 2004 in Hesse. Yearly numbers of severe accidents increased from 194 (1995) to 239 (2004) (**Fig. 6.1**). Hessian hunting statistics show a similar trend with increasing numbers of roe deer and wild boar killed by collisions with cars between 1960 and 2003 (**Fig. 6.2**).

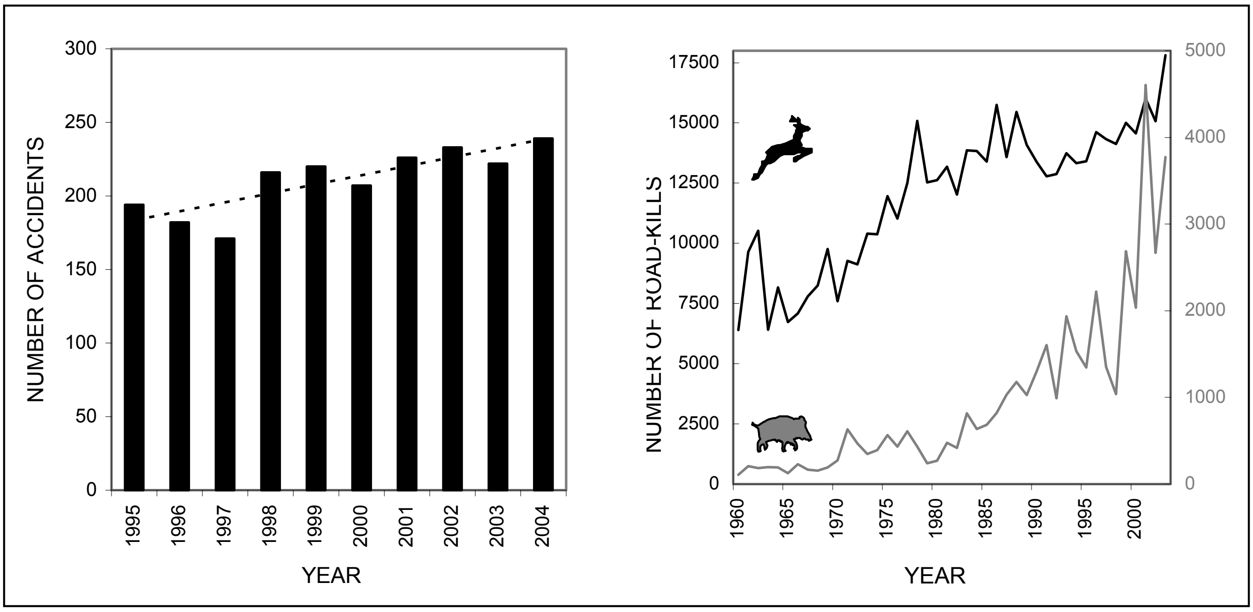


Fig. 6.1 (left): *Numbers of police-reported wildlife-vehicle collisions followed by severe material or personal damage in Hesse (1995-2004).*

Fig. 6.2 (right)*: Numbers of road-killed roe deer and wild boar (1960-2003), based on Hessian hunting statistics (Roedenbeck and Köhler 2006, modified).*

Seasonally, wildlife accidents most frequently occurred between April and November. The highest peak lies in summer months July and August, while two smaller peaks occur in spring (May) and fall (October). During winter (December to March) accident numbers decreased (**Fig. 6.3**).

Most accidents occurred in evening and nighttime hours between 7pm and 8am, with one peak after sunset between 9-11pm and another from 5-7am at sunrise in early morning hours. During daylight from 9am to 6pm vehicle-wildlife accidents were low, although traffic is usually high during daytimes (**Fig. 6.4**).

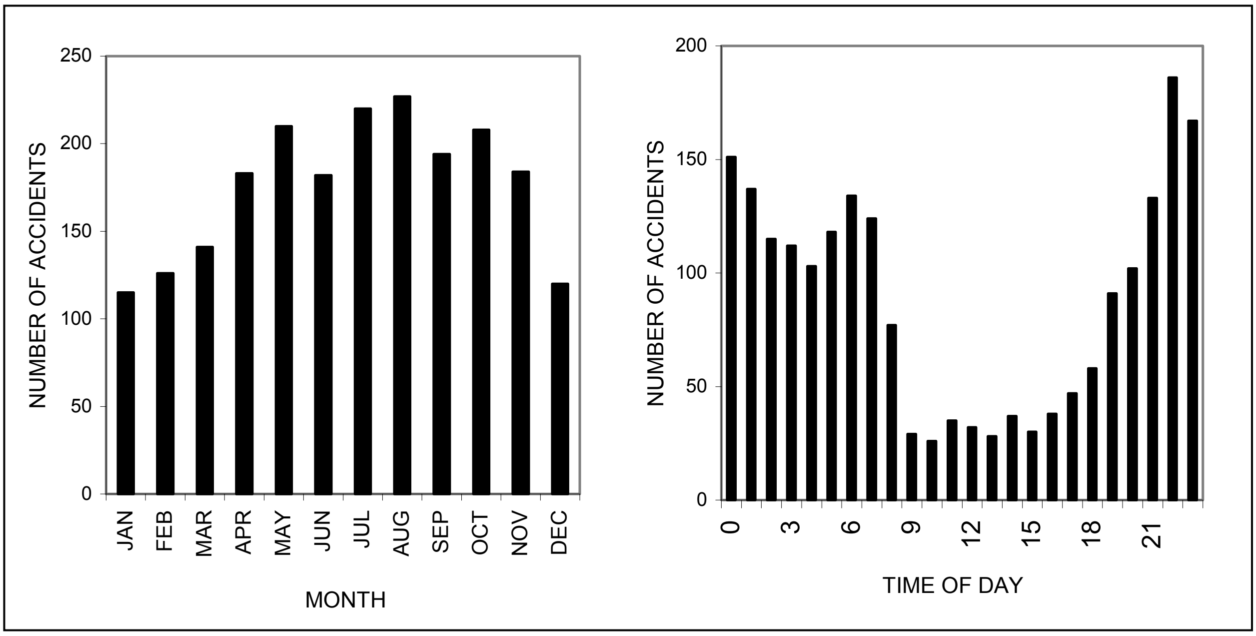


***Fig. 6.3**: Seasonal patterns of wildlife accidents in Hesse, followed by severe material or personal damage (N= 2,111; 1995-2004).*

***Fig. 6.4**: Daily patterns of wildlife accidents in Hesse, followed by severe material or personal damage (N= 2,111; 1995-2004).*

6.3.2 SPATIAL CLUSTERING OF ACCIDENTS

Vehicle-wildlife collisions occurred over almost the entire road network. Following the kernel density estimate there are three hotspots of collisions in the urban areas of Kassel, Giessen, and Frankfurt, where road and traffic densities are highest (**Fig. 6.5**). Examining the K-function for the distribution of fatalities along highways showed that no significant clustering occurred. Observed values of *K(t)* lie almost within the 95% confidence envelope indicating a random distribution of accidents. At scales from 115 to 225km the accidents are not randomly distributed, but tend towards regularity (**Fig. 6.6**).

Fig. 6.5: (A) Distribution of police-reported wildlife-vehicle collision sites followed by severe material or personal damage in a period from 1995 to 2004 in Hesse, Germany. (B) Kernel density estimate of accidents sites.

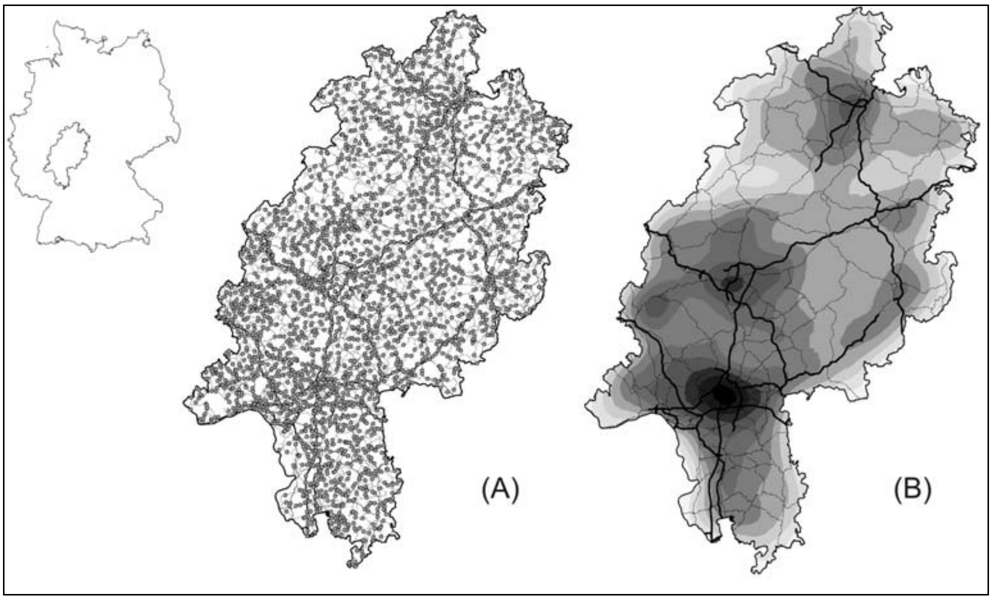
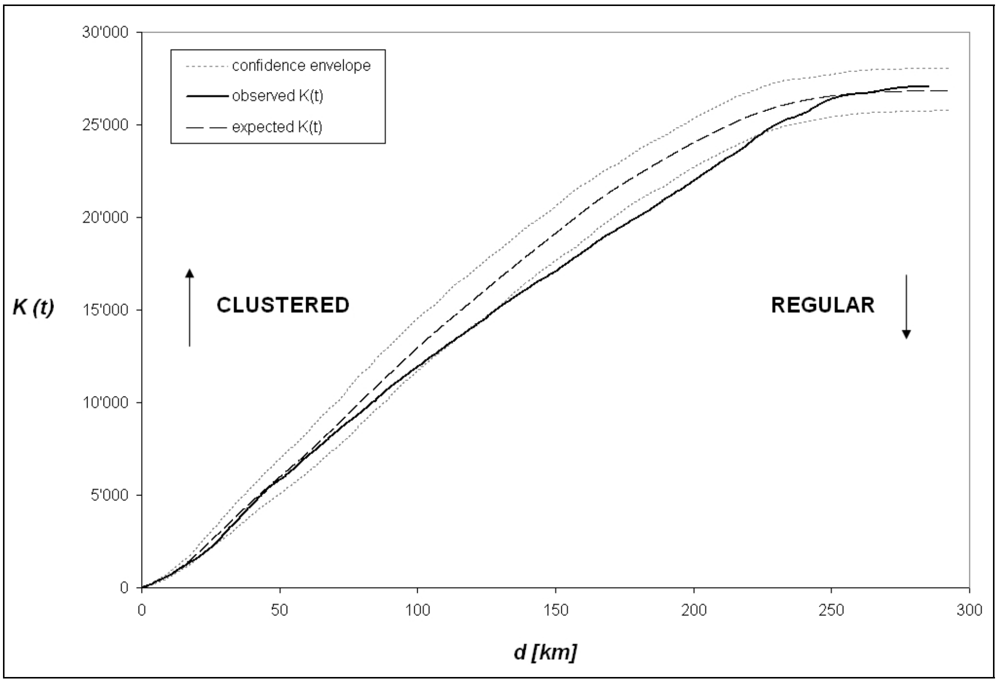


Fig. 6.6: Spatial pattern analysis using the network K-function for describing the distribution of vehicle-wildlife accidents on highways in Hesse. The observed statistic K(t) is plotted against d. The grey lines give the 95% confidence envelope and values of expected K(t).



6.3.3 VARIABLES ASSOCIATED WITH ACCIDENT SITES

Mann-Whitney U-Test and χ^2- test revealed differences between the means of accident and control sites for most of the landscape variables observed. As the sample size was large, we will only refer to variables showing a large or medium-sized effect (**Tab. 6.4**).

Accident sites had a significantly higher amount of forest, and forest patches were larger than in control areas. Accordingly, accidents took place in closer proximity to forest edges and in larger distances to urban areas. The amount of urban area, road and railroad density was lower in accident sites than in control sites.

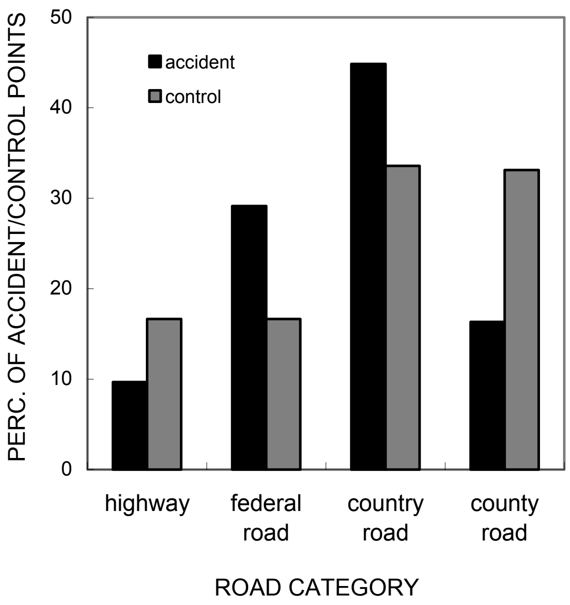
Accident sites occurred more often where roads crossed large non-fragmented areas, and where both roadsides differed, i.e. where forest patches and open areas were separated by roads, whereas control sites showed a higher similarity of road sides.

Tab. 6.4*: Descriptive statistics for variables determining presence or absence of vehicle-wildlife collisions. Mean, maximum values and standard error of variables are shown along with results of Mann-Whitney U-Test and χ^2- test. Variable coding is explained in* ***Tab. 6.3****. N= 1,726 accident sites, and 1,726 control sites.*

Categorical variables		Control sites			Accident sites			U-Test		Effect size
		max	mean	SD	max	mean	SD	Z	P-value	Cohen d
FOREST$_{PROP}$	[%]	100	24.19	28.79	100	41.11	33.18	-16.062	<0.001	0.545
OPEN$_{PROP}$	[%]	100	50.22	30.98	100	48.83	30.91	-1.428	0.153	-0.045
DECID$_{PROP}$	[%]	97.16	7.47	12.04	86.37	11.12	14.52	-8.592	<0.001	0.274
CONIFER$_{PROP}$	[%]	95.63	6.13	14.06	99.88	9.52	17.27	-9.889	<0.001	0.215
AGRIC$_{PROP}$	[%]	99.65	30.77	27.65	100	31.00	28.19	-0.186	0.852	0.008
FOREST$_{DIST}$	[m]	1493.91	167.84	218.12	1517.45	88.53	164.53	-13.732	<0.001	-0.411
FOREST$_{ED}$	[m/m²]	13.26	3.06	2.31	12.68	3.38	2.23	-4.720	<0.001	0.141
MSI	-	5.27	1.56	0.42	3.97	1.48	0.39	-5.679	<0.001	-0.196
MPS	[ha]	78.14	9.67	16.42	78.54	19.05	23.40	-15.161	<0.001	0.464
ROAD$_{KM}$	[km]	13.88	1.97	1.39	11.15	1.57	0.99	-11.909	<0.001	-0.330
TRAFFIC	[cars/d]	154.84	16.89	28.08	148.13	11.53	19.76	-0.679	0.497	-0.221
URBAN$_{PROP}$	[%]	98.83	16.78	22.58	89.46	5.22	10.66	-17.963	<0.001	-0.655
URBAN$_{DIST}$	[m]	4466.06	446.96	527.94	3427.25	705.99	544.66	-17.481	<0.001	0.483
UNFRAG	[km²]	130.93	14.94	17.62	141.42	22.68	21.49	-14.573	<0.001	0.394
TRAIN$_{M}$	[m]	8149.02	298.41	720.03	6695.1	165.22	430.77	-5.324	<0.001	-0.224
CAPREOLUS	[#/Km²]	3.57	2.40	0.79	3.57	2.53	0.62	-3.029	0.002	0.184
SUSSCROFA	[#/Km²]	2.75	1.52	0.62	2.75	1.69	0.60	-6.935	<0.001	0.278
CORR$_{DIST}$	[km]	35.29	7.06	6.28	34.94	6.25	6.49	-5.307	<0.001	-0.127
FFH$_{DIST}$	[km]	8.14	1.66	1.46	8.06	1.55	1.47	-2.804	0.005	-0.075
LAKE$_{PROP}$	[%]	41.52	0.27	2.12	47.15	0.30	1.98	-1.037	0.300	0.015
WATER$_{KM}$	[km]	6.07	1.20	1.03	8.21	1.18	0.96	-0.079	0.937	-0.020
WATER$_{DIST}$	[m]	2454.36	269.79	293.95	1927.67	255.34	269.55	-1.040	0.299	-0.051
CURVE$_{ID}$	[km]	2.67	1.06	0.11	2.11	1.04	0.09	-8.474	<0.001	-0.193
ASL$_{RANGE}$	[m]	275.06	60.85	41.45	247.89	69.43	41.77	-6.548	<0.001	0.206
SIMILAR$_{ID}$	-	100	15.52	19.38	100	21.08	22.08	-8.524	<0.001	0.268
Categorical variables								**x^2- test**		
								x^2	**P-value**	
ROAD$_{CAT}$								216.794	<0.001	
WATER$_{AVAIL}$								29.974	<0.001	

With respect to road characteristics accident sites showed a larger variation in topography than control sites. Wildlife accidents were more common on federal roads (`Bundesstrasse´) and country roads (`Landesstrasse´) and less common on highways (`Autobahn´), and small county roads (`Kreisstrasse´) (**Fig. 6.7**). Accident numbers increase with increasing traffic density, peak at approximately 2,000 cars per day, and start decreasing continuously at higher traffic densities (**Fig. 6.8**).

Accident sites and control sites were similar with regard to the amount of agricultural and open land, the amount of lakes, density of water courses and distance to water. Also, effect sizes of road curvature, population size, and the distance to protected areas and wildlife corridors were too small to make inferences. Many parameters showed a high standard error indicating a huge variability of parameter values at both accident and control sites (**Tab. 6.4**).



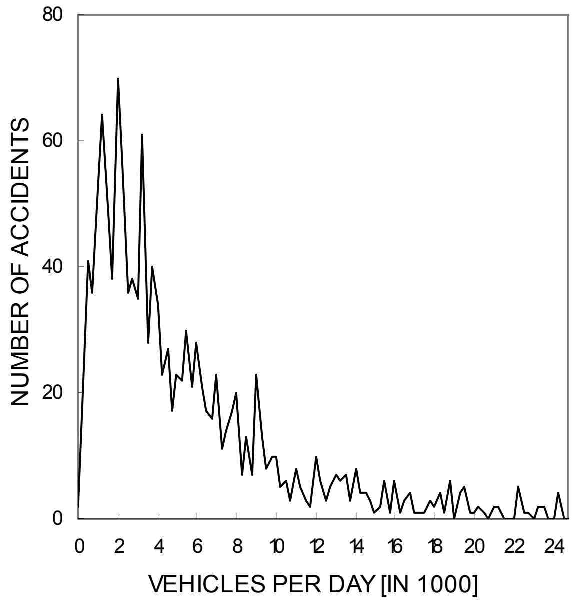


***Fig. 6.7**: Percentage of accidents and control sites for different road categories in Hesse (n=1,726 accident and 1,726 control sites).*

***Fig. 6.8**: Relationship between traffic volume and numbers of wildlife-vehicle collisions during 1995 and 2004 in Hesse (Germany). Traffic volumes were grouped into 50 classes (N= 925). We cut the graph at 25,000 cars per day, as the curve levels. Maximum traffic densities in the study area reached 154,000 cars per day.*

6.3.4 PREDICTIVE MODELS

The models under investigation had a very different predictive capacity. The ROAD model ranked highest among the four models (w_i= 97.4%), including percentage of forest and urban area, road category, variation in topography, road curvature, road and traffic density (**Tab. 6.5**). Despite the small number of variables included, this model correctly classified 65.4% of the observations, 81.7% of the accident sites and 45% of the control sites.

The predictions made by the GLOBAL model (**Tab. 6.6**) correctly classified 70.5% of the observations, however, it had a very low relative model weight (w_i= 0%), because of the large number of included variables. The simplified FACTOR model correctly classified 65.6% of the observations and had a better relative model weight, because of the reduced number of included variables (w_i = 6.3%). The HABITAT model had a low relative model weight (w_i = 0%) and the lowest predictive capacity of all four models (**Tab. 6.7**).

***Tab. 6.5**: Coefficients (B) of the binary logistic model (ROAD model) fitted to the observations of points with and without wildlife-vehicle collisions. Model coefficients are shown along with their standard errors and Wald- significance tests.*

	Coefficient(B)	SE	Wald- χ^2	df	P-value
URBAN$_{PROP}$	-0.036	0.004	102.395	1	<0.001
FOREST$_{PROP}$	0.010	0.002	43.938	1	<0.001
ROAD$_{CAT}$	-1.061	0.218	23.627	1	<0.001
ASLR$_{ANGE}$	-0.003	0.001	7.930	1	0.005
CURVE$_{ID}$	-1.225	0.474	6.663	1	0.010
ROAD$_{KM}$	-0.039	0.046	0.734	1	0.392
TRAFFIC	0.002	0.003	0.328	1	0.567
constant	*1.894*	*0.490*	*14.951*	*1*	*<0.001*

***Tab. 6.6**: Coefficients (B) of the binary logistic model (GLOBAL model) fitted to the observations of points with and without wildlife-vehicle collisions. Model coefficients are shown along with their standard errors and Wald- significance tests.*

	Coefficient(B)	SE	Wald- χ^2	df	P-value
ROAD$_{CAT}$	-1.019	0.154	43.820	1	<0.001
ASL$_{RANGE}$	-0,008	0.001	36.504	1	<0.001
CAPREOLUS	-3.344	0.090	14.599	1	<0.001
URBANPROP	-0.020	0.005	13.758	1	<0.001
UNFRAG	0.010	0.003	12.837	1	<0.001
SUSSCROFA	0.336	0.095	12.444	1	<0.001
CURVE$_{ID}$	-1.515	0.469	10.456	1	0.001
CONIFER$_{PROP}$	-0.011	0.004	9.201	1	0.002
FOREST$_{PROP}$	0.011	0.004	8.639	1	0.003
FOREST$_{DIST}$	-0.002	0.001	8.564	1	0.003
CORR$_{DIST}$	0.020	0.008	6.606	1	0.010
ROAD$_{KM}$	-0.095	0.048	4.031	1	0.045
SIMILAR$_{ID}$	0.004	0.002	3.483	1	0.062
WATER$_{AVAIL}$	-0.177	0.101	3.038	1	0.081
FFH$_{DIST}$	0.041	0.030	1.844	1	0.175
MSI	0.106	0.133	0.631	1	0.427
DECID$_{PROP}$	-0.003	0.004	0.491	1	0.483
URBAN$_{DIST}$	0.00001	0.000	0.365	1	0.546
TRAIN$_{M}$	0.00005	0.000	0.348	1	0.555
MPS	-0.002	0.005	0.239	1	0.625
FOREST$_{ED}$	0.014	0.032	0.198	1	0.657
constant	*1.894*	*0.490*	*14.951*	*1*	*0.000*

***Tab. 6.7**: Ranking of four models of landscape factors influencing wildlife-vehicle accidents in Hesse. Rankings are based on Akaike´s Information Criterion (AIC), and relative model weights are indicated by Akaike weights (ω_i).*

Model	Concordance			*AIC*	*Δi*	*ωi*
	accident	control	total			
ROAD model	81.7	45.0	65.4	3265.23	0.000	0.974
FACTOR model	81.5	45.6	65.6	3272.51	7.28	0.026
GLOBAL model	81.3	55.8	70.5	3351.89	86.66	0.000
HABITAT model	62.2	63.7	63.0	3902.92	637.69	0.000

Model parameter:

[1] FOREST$_{PROP}$, TRAFFIC, ROAD$_{KM}$, ROAD$_{CAT}$, URBAN$_{PROP}$, CURVE$_{ID}$, ASL$_{RANGE}$

[2] FOREST$_{PROP}$, OPEN$_{PROP}$, TRAFFIC, HIGHWAY$_{ID}$, URBAN$_{DIST}$, UNFRAG, CAPREOLUS, CORR$_{DIST}$, WATER$_{KM}$, CURVE$_{ID}$, SIMILAR$_{ID}$

[3] The global model included all variables distinguishing significantly between accident and control sites following U- and χ^2- test (see Tab 2).

[4] FOREST$_{PROP}$, FOREST$_{DIST}$, FOREST$_{ED}$, MSI, MPS, UNFRAG, CAPREOLUS, SUSSCROFA, FFH$_{DIST}$, CORR$_{DIST}$, SIMILAR$_{ID}$

6.3.5 Model validation

When applying the four models to validation sites not included in model composition, the comparison of ROC curves and asymptotic significances showed that all models succeeded in distinguishing between accident and control sites (**Tab. 6.8**). Based on their distances from the reference line, all four models were doing better than guessing (**Fig. 6.9**). The area under the curve was highest for the GLOBAL model (*AUC*=0.76), showing an overall concordance of 72.5%, and correctly classifying 69% of all accident sites, and 76% of all control sites. The other three models were also significant producing overall concordances of 70.2% (FACTOR model), 69.4% (ROAD model), and 67.1% (HABITAT model). However, though doing well in classifying control sites, they showed rather small correct classification rates for accident sites: 62.7% (FACTOR model), 53.2% (ROAD model), and 54.0% (HABITAT model) (**Tab. 6.8**).

Tab. 6.8*: Application of four models to 252 randomly selected validation sites in Hesse. Models predict the probability for being a vehicle-wildlife collision or control site. For model accuracy we calculated the area under the ROC curve (AUC) (see* ***Fig. 6.9****). Sensitivity and specificity indicate the amount of correct classifications. We set the cutoff-value at a maximum of correct classifications (cutoff=* $\sum_{max}$ *sensitivity + specificity).*

	GLOBAL model	FACTOR model	ROAD model	HABITAT model
AUC	0.759	0.739	0.721	0.719
significance	<0.001	<0.001	<0.001	<0.001
Cutoff-value	57.6	40.72	63.97	55.88
Sensitivity (% correct accidents)	69.0	62.7	53.2	54.0
Specificity (% correct control)	76.0	77.6	86.4	80.0
Overall concordance with actual status (%)	72.5	70.2	69.4	67.06

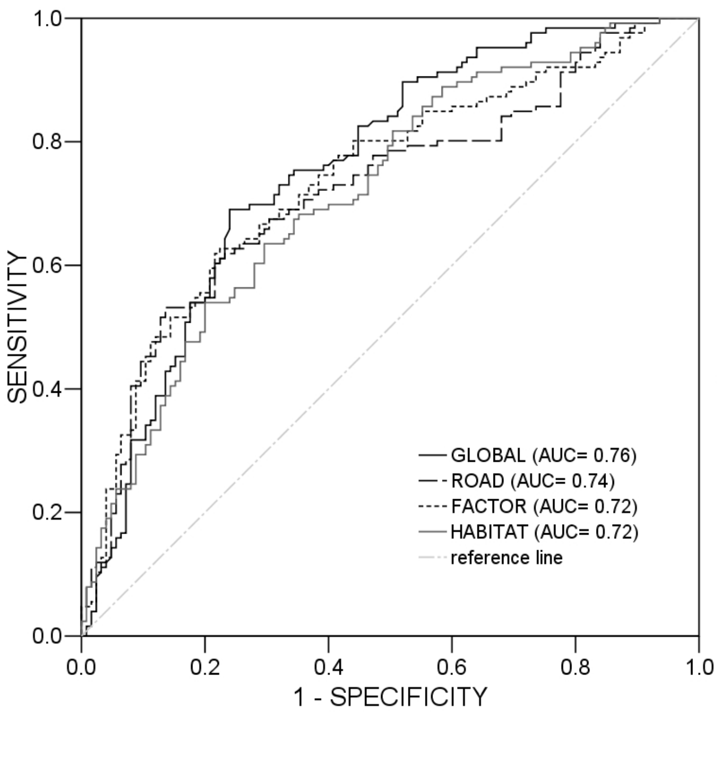


Fig. 6.9*: ROC curves for four models (see* ***Tab. 6.7****) distinguishing between accident and control sites, based on 252 validation sites not included into model building. Coordinates are based on the rate of correct classifications of accident sites (sensivity) against the rate of misclassifications of control sites (1 - specifity). The area under the curve (AUC) is an index for model performance (see* ***Tab. 6.8****).*

6.4 DISCUSSION

6.4.1 TEMPORAL PATTERNS

Temporally peaks of wildlife accidents during darkness, with one peak 1-2 hours after sunset and another at sunrise are known from other studies (Allen and McCullough 1976; Hartwig 1993; Joyce and Mahoney 2001) (**Tab. 6.1** and **Tab. 6.2**). Since vehicle traffic is usually low during night, this pattern is a result of animal behaviour. Roe deer and wild boar move from woods to forage areas of the open landscape at dusk and retreat to the forested slopes at dawn, as shown for white-tailed deer (Carbough et al. 1975).

The seasonal pattern we found, with three monthly peaks of collisions in July/August, spring and fall, is also known for roe-deer in Denmark. However, spring and summer peaks are a bit earlier in Denmark and the highest peak is in fall and not in summer (Madsen et al. 2002). The seasonal pattern seems to be associated with wildlife dispersal and breeding behaviour, corresponding to spring dispersal of young males in May and rutting activity in July and August.

The peak in fall may be a result of on overlap between wildlife and human activity patterns, as wildlife activity in dusk coincides with rush-hour traffic. Also, weather and visibility conditions are poor in fall decreasing the ability to stop on the road when an animal approaches. Rutting season of wild boar in winter is not reflected in the data.

In Pennsylvania and Iowa, white-tailed deer show no summer peak, but increases in accident numbers in May and November as a result of fall hunting seasons and later rutting seasons (Bellis and Graves 1971; Puglisi et al. 1974; Hubbard et al. 2000). Joyce and Mahoney (2001) associate a summer peak of moose accidents in Newfoundland with holiday traffic.

6.4.2 SPATIAL PATTERNS

We found two important road characteristics influencing the risk of collisions: road category and traffic intensity. First, collisions were more common on federal and country roads than on highways and small county roads. This is basically a consequence of traffic network characteristics, as federal roads are more frequent than highways. This pattern also seems to be a consequence of differing traffic densities which are higher on highways. The relationship between traffic density and accidents numbers is not linear. From a given threshold value animals seem to avoid road crossings and high traffic densities act as movement barriers. The same pattern has been shown for moose in Sweden, and the peak found by Seiler (2005) is comparable to the peak found in our analysis at about 2,000 cars per day. Telemetry studies for roe-deer in Denmark support our findings, showing that roads with rather high traffic volume may constitute natural home-range boundaries (Jeppesen 1990), and as such, movement barriers may limit genetic exchange. This result deserves special attention when designing mitigation measures. Though the construction of fences may decrease collisions and road mortality, fences increase the barrier effect of highways, and may decrease population viability, if not combined with safe passageways.

Habitat parameters influencing high collision risks were mainly forest-related. Roe deer and wild boar live in forest habitats, use wooden patches as hiding place and avoid areas with human activity. In forest areas, deer use the rights-of-way for feeding and face a higher risk of collisions, in contrast to agricultural areas, where they feed in agricultural land further away from the road (Carbough et al. 1975). Also, woody cover may hide deer from motorists´ sight, consequently shortening reaction time for a driver to avoid collisions (Madsen et al. 2002). The same characteristics as found here were shown for roe deer and wild boar in Spain (Malo et al. 2004), for white-tailed deer in Pennsylvania (Bashore et al. 1985) and Illinois (Finder et al. 1999), and for moose in Sweden (Seiler 2005). Only Nielsen et al. (2003) excluded forest related variables from his models, but his study was conducted in an urban area.

Referring to habitat preferences a hotspot for collisions are woodland-field intersections. These habitat borders are often frequented by deer, which live in the forest but forage on open lands (Carbough et al. 1975; Waring et al. 1991). While feeding, they often remain close to the wood and sometimes prefer green vegetation on forest edges (Puglisi 1974; Bashore et al. 1985; Finder et al. 1999). High usage frequencies lead to higher risks of collisions.

Other habitat variables increasing the risk of collisions in Hesse, as the size of non-fragmented area around the accident site, enhance habitat quality for forest species. Our hypothesis is supported by Malo et al. (2004) for roe deer and wild boar in Spain, and by findings for white-tailed deer in the USA (Finder et al. 1999; Nielsen et al. 2002), where more accidents occurred in areas with high landscape diversity and recreational land, inhabited by larger populations.

6.5 CONCLUSIONS AND IMPLICATIONS

All of our models succeeded in distinguishing between accident and control sites. However, we recommend only using the GLOBAL model for management purposes, because validation indicated the highest prediction performance for this model. The ROAD model was selected as best model according to simplicity and parsimony, because the lowest number of parameters was necessary to make acceptable predictions. However, this model - likewise the FACTOR and HABITAT model - showed high misclassification rates in validation, especially for accident sites. When making recommendations for locations of mitigation measures a misclassification of accidents sites is much more severe than misclassifying control sites. Therefore, none of the three models is precise enough to be used for mitigation purposes. Of course, a simpler model based on fewer numbers of landscape variables would have been desirable. However, since all data required for the GLOBAL model are readily available, an application in road offices is nonetheless possible.

Reasons for high misclassification rates might be (i) the pooling of species, (ii) data quality, and (iii) the ignorance of relevant parameters during variable measurement and model composition.

First, pooling of species was necessary, because the data set does not give information on the species. However, we think that is not a major problem, as roe deer and wild boar show comparable habitat use and area requirements. Other authors have designed models pooling roe deer, wild boar, and red deer, and achieved a rather high model performance of 87% of correct predictions (Malo et al. 2005).

Second, the data we used did not include all collisions between cars and wildlife in Hesse, but only those followed by severe damage. The lack of the majority of accidents followed by little damage to the car but death for the animal may cause some error in model composition. Consequently, control sites are associated with more uncertainty than accident sites, as collisions followed by small property damage may have occurred at control sites or may occur in the future. However, mitigation efforts should focus on accident hotspots, and the rate of correct classifications of accident sites is quite reliable in the GLOBAL model.

Third, possible parameters influencing vehicle-wildlife accidents were not included in our analysis. For example, specific crop types and the quality and density of vegetation have been proven to have a significant influence on white-tailed deer and roe deer accidents in Pennsylvania and Denmark (Carbough et al. 1975; Madsen et al. 2002). Presence of guardrails and mitigation measures such as deer crossing signs may also be of special importance (Malo et al. 2004; Seiler 2005). For example, a study including information on the condition of mitigation measures showed that increasing the number of fully repaired fences reduced the probability of white-tailed deer accidents (Bashore et al. 1985). This study was based on detailed field surveys, included detailed local scale information and achieved an extraordinary model fit of 90% of correct classifications.

We conclude that the lack of information about other potential impact factors is the most important point with respect to model performance. To achieve higher model fits, more detailed information is needed, such as data about the distribution of fences, guardrails and warning sings. Hence, to infer explicit recommendations about where to build up a specific mitigation measure in a concrete situation, we recommend collecting detailed local scale data for the given circumstances, including interviews with local huntsmen and experts.

At large scales, such data collection is too time and cost intensive. However, decision support is urgently required at large scales, especially in the context of defragmentation programmes. For large scale purposes, model fit of our GLOBAL model, with 72.5% of correct classifications, is quite reliable and comparable to other studies about white-tailed deer in the USA, and moose in Sweden, with overall concordances of 63% (Hubbard et al. 2000), 77% (Nielsen et al. 2003), and 78% (Seiler 2005) (**Tab. 6.8**). Therefore, we recommend using our GLOBAL model at large scales in the context of de-fragmentation programs. For example, in the Netherlands a coordinated national long-term de-fragmentation Programme was initiated in 2001, developing solutions for problem spots and prioritising actions (van der Grift 2005). Within this framework, scientists identified spots where de-fragmentation measures at transport corridors are most urgent, and mitigation measures in 14 priority areas will be constructed by 2010. Within the Dutch approach de-fragmentation measures are based on the expected increase of population viability due to potential crossing structures. In contrast, the study in hand is based on a more anthropocentric perspective aiming at mitigating vehicle-wildlife accidents and recommending mitigation measures at collision spots. However, both approaches are reasonable and aid one another to address hotspots for mitigation efforts.

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Chapter 7.
Landscape-scale Effects of Roads on Wildlife

Discussion, Implications and Conclusions

This chapter will be a general discussion of the research results presented in chapters 2-6. First, we will give a short summary of each papers´ results. Then, we will move to a more general discussion of the relevance and significance of each paper in the practical context of political decision making.

7.1 DISCUSSION CHAPTER 2

THE RAUISCHHOLZHAUSEN AGENDA FOR ROAD ECOLOGY

In chapter 2 we address the discrepancy between scientific research and decision-making in road ecology. Although there is a growing body of evidence of negative impacts of roads on wildlife (Forman et al. 2002, Sherwood et al. 2002, Spellerberg 2002, Trombulak and Frissell 2000, Underhill and Angold 2000,) these results have had comparatively little effect on transportation planning (OECD 2002, UBA 2003). We argue that this relates to the nature of former road research, because (a) the questions addressed by road ecologists were not directly relevant to the practical issues of road planning and construction; and (b) most road studies failed to have high evidentiary weight. Reasons for low evidentiary weight were mainly low-quality study designs producing contradictory results and recommendations. This creates uncertainty and leads to considerable effort being expended to "resolve" apparently contradictory results. Study designs of low inferential strength weaken the reliance in science with the overall result of a constantly growing road network, where economic and socially driven needs far outweigh uncertain ecological arguments.

We argue that road research could increase its relevance, influence and effectiveness by (a) addressing questions of direct management concern, and (b) designing studies that have high inferential strength. Consequently, we first identify five questions of most direct relevance to the decision process. Then, we develop study designs having high evidentiary weight for any of the research questions outlined. We argue that the BACI design is the study design with highest inferential strength, where impact sites affected by a road (I) are compared with non-affected control sites (C) both before (B) and after (A) some intervention. A well designed BACI with replicated sites and replicated sampling allows drawing conclusions about the effect of roads on wildlife. We are aware of the reality that a well designed BACI is not possible in any situation. Consequently we present two alternative designs for any question of higher feasibility but lower inferential strength. The core of the first paper is a quality standard for future research. The "Agenda" (**Fig. 2.1**) outlines three designs for any research question and giving a recommendation about which study should be prosecuted based on its feasibility and inferential strength.

In the context of this thesis and the following papers we want to pay special attention to one particular research question outlined in chapter 2. This is research question 5 asking: Under which circumstances do road networks affect population persistence at the landscape scale? This question addresses the importance of scale in road ecology. We argue that answers about the landscape-scale effects of roads are urgently required, because the most pressing policy and management issues are generally not at the local but at the landscape scales (National Research Council 2005). However, looking at the research Agenda we see that the inferential strength of all feasible studies at the landscape scale will be comparatively low. Consequently, it is inevitable that the uncertainty associated with any conclusion at the landscape-scale will be high. Reasons for this are manifold.

First, at large scales the researcher can hardly sample the environmental variable of interest by himself. Consequently, landscape-scale analyses require wildlife monitoring data being readily available. However, such data rarely exists. And if it exists the researcher has had no influence on experimental manipulation. He cannot decide which species to be observed, he cannot determine which parameters to be sampled, and he has no influence on data quality. Large-scale samplings are often conducted by a range of unskilled volunteers, and the resulting density maps of species truly reflect the distribution of experience in sampling. In addition to this, such mappings rarely show full area coverage, and dealing with spatial gaps in sampling is problematic. It is also problematic that the boundaries of the landscapes observed are often determined by political concerns (e.g. administrative districts), and do not follow population distributions or home ranges of the species observed. The environmental variable measured is often just a substitute for population abundances. For example, the amount of animals hunted is a used as a representative for abundances, although harvest does also reflect hunting intensity. In summary, low quality of wildlife data reduces inferential strength of landscape-scale studies.

Second, at large scales the before sampling of a full BACI is seldom possible, because sampling requires extensive time and labour. The studies prosecuted are truly CI designs, having much lower inferential strength than BACIs. In addition to this, populations can almost never be observed over long-terms, and short-term studies tell us nothing about possible lagged effects.

Third, at large scales choice of impact and control sites is problematic. The base unit of a landscape-scale study is a non-overlapping landscape. Landscapes should be replicated to measure the environmental variable of interest (e.g. population abundance) across different road densities. It is crucial that in all landscapes all other potential impact factors on population abundances are comparable. But it is hardly achievable to find several non-overlapping landscapes where all other impact factors (e.g. intensity of agricultural use, density of settlements) are the same. So, the researcher cannot rule out that a difference in population abundances observed was due to another impact than road network density. Choosing highly comparable sites will inevitably reduce sample size, and this loss of replication reduces inferential strength of a study.

Following this compilation we see that landscape-scale studies cannot stick to the quality standard suggested by the Agenda. A manipulative BACI-design with highest possible inferential strength will not be feasible at large scales. Methodological problems and problems of data quality will always lead to results of low inferential strength. It is important to internalize this fundamental idea, because a multitude of the problems and consequences theoretically described in chapter 2 appeared in the empirical analysis pursued in the framework of this thesis (see chapter 4 and 5).

So what can be done? The answers needed for decision making cannot be obtained by extrapolating local results to the landscape scale, as this leads to various extrapolation problems (see chapter 1). In fact, we have to conduct the analysis at the scales where answers are required. In doing so, studies have to be conducted with the best study design feasible under the given circumstances and the given data qualities, and results have to be interpreted with adequate caution. If these basic requirements are fulfilled, the results will be the best we can get at large scales, and decision makers must not demand better proofs before incorporating the results into their decision making.

7.2 DISCUSSION CHAPTER 3

LANDSCAPE FRAGMENTATION IN HESSE

In chapter 3 we show the development of landscape fragmentation in the federal state of Hesse in Germany between 1930 and 2002. The results serve as a base for continuative analysis about the effects of roads on wildlife populations (see chapter 4). We used index effective mesh size (m_{eff}) (Jaeger 2000) to measure the degree of fragmentation. We conducted the analysis in six time steps for three different spatial scales: the three regional boards of Hesse (Regierungspräsidien Kassel, Giessen, and Darmstadt), the 26 administrative districts, and the 59 natural areas. Results show that landscape fragmentation by roads and settlements in Hesse increased continuously since 1930.

There is an increasing amount of quantitative studies illustrating the historical development and current situation of landscape fragmentation in Germany (review in Grau 1998). Former nation wide documentations were conducted by Lassen (1979, 1987), Dosch et al. (1995), Schumacher and Walz (2000), and Gawlak (2001). An old state wide monitoring was conducted in North-Rhine Westphalia (Baumann and Hinterlang 2001). These studies were not comparable, because landscape fragmentation had been measured with different indices. The Federal Nature Conservation Agency (BfN), for instance, used the number of undissected low traffic regions larger than 100 square kilometres (UZVR) (BfN 1999). And a growing number of different indices has been used in literature, such as traffic lane density, landscape dissection index (Bowen and Burgess 1981), and the relative portioning index used by the Federal Statistical Office (Deggau et al. 1992).

Jaeger (2000) developed the first well accepted index, effective mesh size (m_{eff}), which is scientifically confirmed. A systematic comparison with six other measures (number of undissected areas n, average size of areas F, landscape dissection index LDI, and the relative portioning index PI_{rel}) with respect to nine suitability criteria showed that m_{eff} has several advantages, and is well suited for comparing the fragmentation of regions with differing total size. In contrast to the undissected area index (UZVR), m_{eff} responds to the structure of a landscape, and not only to patches larger than 100 square kilometres. Another advantage of m_{eff} is that it is based on an ecological process, because it indicates the probability that two individuals of one species meet in a landscape fragmented by barrier elements such as roads. These and other advantages, like its mathematical simplicity, leaded to the state wide application of m_{eff} in environmental monitoring. Following its first application in the Swiss area "Raum Kreuzung Schweizer Mittelland" (Müller et al. 1998), and in the German region „Strohgäu" in Baden-Württemberg (Jaeger 1999), the index has been used many times. Numerous federal states conducted state wide documentations of the current situation, such as Baden-Wuerttemberg (Jaeger et al 2001, Esswein et al 2002), Hesse (Roedenbeck 2005, Roedenbeck et al. 2005), Saxony (LfUG Sachsen 2002), Bavaria (Esswein and Schwarz von Raumer 2003, Esswein et al. 2004), Schleswig-Holstein (Neumann-Finke 2004) and Thuringia (TLUG 2004). There are also applications in smaller sub-areas (Clausing 2006, Voerkel 2005, Walz 2005), and further enhancements of the index for particular circumstances (Moser et al. 2007, Penn-Bressel 2005, Schwarz von Raumer et al. 2006, UBA 2003). Furthermore, the index has been applied abroad, e.g., in South America (Baldi et al. 2006), Italy (Padoa-Schioppa 2006), Switzerland (Jaeger et al. 2006, Peter and Meier 2003), Canada and France. A comparison of European countries is planned.

The benefit of such documentations becomes apparent when illuminating the significance of landscape fragmentation as environmental problem in policy (Schupp 2005). Although discussed in circles of experts, landscape fragmentation has not reached significance in policy. One reason is that the increase of road and settlement networks proceeds stealthy, and as such, is underestimated, although a persisting negative trend has been proven. This type of environmental problems has been denominated as "persistent environmental problems" by the Council of Environmental Advisors in Germany (SRU 2002). It has been identified as one of the most important strategic challenges in environmental policy today. Environmental sciences play a decisive role as a central actor in persistent environmental problems, as they can identify needs for action and support the public awareness of the problem (Schupp 2005). Environmental monitoring is an effective instrument in this context (SRU 2002). Quantification makes the problem comprehensible, uncovers historical developments, and enables comparisons of countries. For this reason, the documentation and quantification of landscape fragmentation is the *sine qua non* for political troubleshooting. One essential requirement of environmental monitoring is to apply standardised methodology, and to use a set of homogeneous key indicators. Only a consistent methodology enables comparisons of countries and periodical updates, as shown by the first consistent historical documentations of fragmentation in Baden-Württemberg and Hesse (Roedenbeck et al. 2005).

Following this idea, the environmental authorities of the state cooperated at the Federal State level in the past years, aiming at standardising environmental indicators, and developing a collective set of key indicators (LIKI) (Schupp 2005). Actors finalised to use a combination of UZVR and m_{eff} as indicators for landscape fragmentation in further environmental monitoring in Germany. Furthermore, they agreed on a set of landscape elements to be used in these documentations. These conventions enable a regular update of the fragmentation documentation in the future. Chapter 3 in this thesis is the basis for such a monitoring in Hesse.

Note: Although benefits of this study are obvious, a monitoring of fragmentation is only of documentary value. As long as there is no evidence about landscape-scale effects of road networks on wildlife populations, it will not provide a sound basis for argumentation. In the face of compelling economic and social arguments the densification of the current road network will continue further on, as long as we cannot actually verify effects of road networks on population persistence.

7.3 DISCUSSION CHAPTER 4

EFFECTS OF ROAD NETWORKS ON ROE DEER AND WILD BOAR IN HESSE

In chapter 4 we take up the historical documentation of landscape fragmentation (chapter 3) and link it to hunting statistics of Hessian mammal species. In doing so we aim at: (1) testing whether m_{eff} is able to indicate the state of wildlife populations alongside indicating the degree of fragmentation (indication quality), and (2) analysing whether road networks affect wildlife populations at the landscape scale. Regression analysis on the level of administrative districts, indicate a relationship between the values of the fragmentation index and hunting statistics. With increasing landscape fragmentation road kills increase and population abundances decrease. The relationships are statistically significant for roe deer (*Capreolus caperolus*), wild boar (*Sus scrofa*), fox (*Vulpes vulpes*) and badger (*Meles meles*) populations.

In recent years, a lot of landscape indices have been developed (Rutledge and Miller 2005), for instance, indices being implemented in the GIS-Tool FRAGSTATS (McGarigal and Marks 1995). Landscape indices describe landscape patterns, for example, the spatial arrangement of habitat patches, the degree of fragmentation, the spatial arrangement of roads, and the density of road networks. However, a review of 566 studies using such landscape indices pointed out that many focussed on describing aspects of pattern with little consideration of their relevance to process (Rutlegde and Miller 2006). As such, the use of landscape indices failed the basic purpose of landscape ecology (see Turner 1989). In contrast to this undesirable development, we were able to show a significant relationship between the values of index m_{eff}, population abundances and road kill rates, respectively, and as such, we provided evidence for a linkage between pattern and process. However, our results have to be interpreted with caution for several reasons:

First, our study a typical CI study (see chapter 2). A BACI design with data from before and after was not feasible, because such data does not exist for the study area. As discussed above, a CI design has always lower inferential strength than a BACI.

Second, the use of harvest statistics as an indicator for population abundance is problematic. Harvest statistics do not only reflect population abundances, but also hunting intensity. Especially for small game species it has been discussed that harvest statistics do not indicate the state of a population (Strauss 2000). This may be one reason, why we did not find any significant result for the small game species investigated in Hesse, such as marten (*Martes foina*), racoon (*Procyon lotor*), hare (*Lepus europaeus*), and grey partridge (*Perdix perdix*). However, harvest statistics were the only data being available in Hesse, and as such, the only data pool we could use. This reflects a typical problem of landscape-scale studies, as the researcher has to use available data without having any influence on data quality (see chapter 7.1).

A third problem arose from the peculiarity of the species observed. Populations of roe deer and wild boar increased over the last years for several reasons, and are generally not threatened by landscape fragmentation. Here, as above, we were faced to a typical problem of landscape-scale studies, as the researcher has no influence on the choice of the species observed (see chapter 7.1). However, we use roe deer and wild boar as representatives for other forest species with large home ranges and comparable habitat requirements (e.g. wildcat and lynx), which are of conservation value but not represented in wildlife monitoring data in Hesse.

A fourth problem arose from study design. As discussed above (see chapter 7.1), base units of landscape-scale road ecology studies are non-overlapping landscapes along a gradient of different road network densities with comparable habitat amounts. However, it was impossible to choose a sample size of landscapes with comparable habitat conditions for the study in hand. Harvest statistics were available on the level of administrative districts, and comprised a sample size of 26 base units. It did not make any sense to further decrease such a small sample size by choosing base units with a similar amount of forest habitat. As a consequence of non-comparable base units, index m_{eff} interacted with other habitat factors. A multiple regression analysis demonstrated that m_{eff} correlates to the amount of forest in an administrative district. Following this results, high population abundances in one district could either profit from a large amount of forest, or from a low degree of fragmentation. Consequently, the study design cannot tell us exactly the exclusive effect of roads.

The problems outlined are *sui generis* by no means, but rather typical for landscape-scale studies. We already made this point in chapter 2 and 7.1, and it was really interesting to see how the problems, theoretically assumed above, really came true when linking pattern to process, and conducting a landscape-scale analysis in practise. We concluded that landscape-scale studies have to be interpreted with caution (see chapter 7.1), nonetheless, we can make some inferences on the basis of the analysis in chapter 4: (1) Large undissected forest areas deserve high priority in the protection of wildlife areas (though we do not know, whether roads or forest cause the main effect on populations). (2) Effective mesh size is an appropriate indicator for the situation of large forest species´ populations (though maybe not for species inhabiting open habitats). (3) For animal welfare, as well as economic and human health purposes, effective mitigation strategies have to be developed to counterbalance the increase of vehicle-wildlife collisions caused by an increasing road network. Making inferences about the exclusive effects of roads on population abundances requires continuative analyses based on wildlife data of higher quality (see chapter 5 and 7.4).

7.4 DISCUSSION CHAPTER 5

EFFECTS OF ROAD NETWORKS ON BROWN HARE IN SWITZERLAND

In chapter 5 we investigated the effects of road networks on brown hare populations in Canton Aargau in Switzerland. Two spotlight taxations of the years 2003 and 2005 served as base data for the abundance of hare populations. We studied road effects particularly with regard to (1) spatial distribution, (2) population abundance, and (3) road mortality of brown hare. An analysis based on plots established in varying distances parallel to roads showed that hares avoid the proximity to roads, and prefer large non-fragmented areas in contrast to small isolated patches. Regression analysis on the basis of a 4x4 km raster grid, with road network density as predictor, and population abundance as independent variable, revealed differences between the effects of different road types. Highly frequented highways, federal roads, and main roads have a negative effect on population abundance, field tracks and footways have a positive effect. The vegetation adjacent to field tracks probably matters in the diet spectrum, while highly frequented roads cause disturbance effects. We could not show any effect of road network density on road mortality.

Chapter 5 aimed at investigating landscape-scale effects of roads on wildlife populations just as chapter 4. The replication of the research question served for several purposes. First, we aimed at investigating different species with different habitat requirements. Roe deer and wild boar in Hesse live in forest habitats, while brown hare in

Switzerland is a synanthropic species living in field areas. In contrast to roe deer and wild boar populations in Hesse, which increased over the last decades, brown hare in Switzerland decreased since the eighties, and is listed as a threatened species in Switzerland today.

Second, we aimed at conducting the analysis on another spatial scale with a different gradient in fragmentation. The home range of brown hare is with 30ha quite smaller than the home range of roe deer and wild boar. Necessarily, we had to adjust the study design to the new scale, and preferred a 4x4km raster grid to the use of administrative districts as base units. At raster grid level, the use of fragmentation index m_{eff} did not make sense anymore. The reason is that boundaries of base units have to be defined for the calculation of m_{eff}. These boundaries are incorporated as virtual barriers into the calculation of the index (Esswein et al. 2002). For example, in a raster grid cell of 16km², which is separated by a 4km road into two halves, the 4 km road is included as a barrier into the calculation of m_{eff}, as well as the raster grid boundaries with a total length of 16 km. This mistake inherent in index calculation does not matter at large scales. However, at smaller scales it falsifies results. Consequently, we used two alternative predictors for describing the degree of fragmentation: the density of roads per raster grid, and the area size of the undissected field inhabited by the hare.

Third, we aimed at using high quality wildlife data. We already discussed the problems of interpreting hunting statistics above, which cannot be used as an index for population abundance as they also reflect hunting intensity (see chapter 7.3). However, in Canton Aargau spotlight taxations were available for brown hare in addition to harvest statistics. These taxations were conducted twice a year, enabling a replication of the study design. Hare locations were mapped spatially explicit allowing a choice of the base units´ boundaries, instead of using determined boundaries such as administrative districts.

As a consequence of the smaller scale and the free choice of base units boundaries we were able to use a larger sample size (N=200 raster grid cells instead of 26 districts in chapter 4). Accordingly, we were able to extract base units with a comparable amount of field habitat. Whereas we were not able to determine whether large wildlife populations in Hesse existed due to forest area or road network density, respectively, we can make inferences about the exclusive effects of roads in Switzerland, because we factored out habitat availability.

Summa summarum: On the basis of the analysis in chapter 5 we can infer that landscape fragmentation affects brown hare populations. Our results are of specific meaning and importance for conservation of brown hare populations. Though a threat caused by landscape fragmentation has been assumed long ago, the effect has not been demonstrated until now. The result is of much higher inferential strength than the former analysis in Hesse (chapter 4). However, this result does not reduce the value of the Hessian analysis. As said in chapter 7.1, adequate study designs are feasible where high quality wildlife data is available. Area-wide taxations such as the spotlight taxation for brown hare in Canton Aargau are rare in Germany and non-existent in Hesse.

7.5 DISCUSSION CHAPTER 6

PREDICTING LOCATIONS OF VEHICLE-WILDLIFE COLLISIONS

In chapter 6 we developed models predicting the risk of vehicle collisions with roe deer and wild boar in Hesse. We used data of 1,726 registered accidents sites (1995-2004), and compared accident locations with a comparable amount of control sites randomly distributed on the entire road network. To describe landscape adjacent to accident and control sites we calculated 27 landscape variables within a 500m radius of each site. The analysis was based on GIS data being readily available in road offices. Results show that hotspots for collisions were woodland-field interfaces frequented by animals when changing between forest habitat and forage areas on open lands. Accident sites occurred where roads crossed large non-fragmented areas. With respect to road characteristics accidents occur at intermediate traffic densities, and more often on federal and country roads than on highways (**Fig. 7.1**). By means of logistic regression analysis we developed four models based on different variable combinations. All models succeeded in predicting vehicle-wildlife accidents. Our GLOBAL model, including all parameters significantly distinguishing between accident and control sites, achieved best model performance in validation.

Chapter 6 is based on the findings of chapter 3 and 4. We demonstrated that landscape fragmentation increased seriously in Hesse since 1930 (chapter 3). Contemporaneously, as a consequence of this development, road mortality of roe deer and wild boar increased (chapter 4). This seems to be no threat to population persistence, because roe deer and wild boar populations increased over time. However, collisions between vehicles and large wildlife species are a serious problem, because of animal welfare purposes, economic and traffic safety reasons (Groot Bruindering and Hazebroek 1996). Road kill numbers cause enormous material costs and human injury. Animal welfare is also relevant, as numerous individuals are injured in traffic and suffer when dying later in habitat further away from the road. Finally, road kills of roe deer and wild boar highlight the situation of other mammal species with comparable habitat and home range requirements. For example, there are efforts to reintroduce lynx and wildcat in Germany (Schadt et al. 2002). When looking at our results for roe deer and wild boar, the question arises as to whether a reintroduction might be successful in the face of the high risk for being killed on the road.

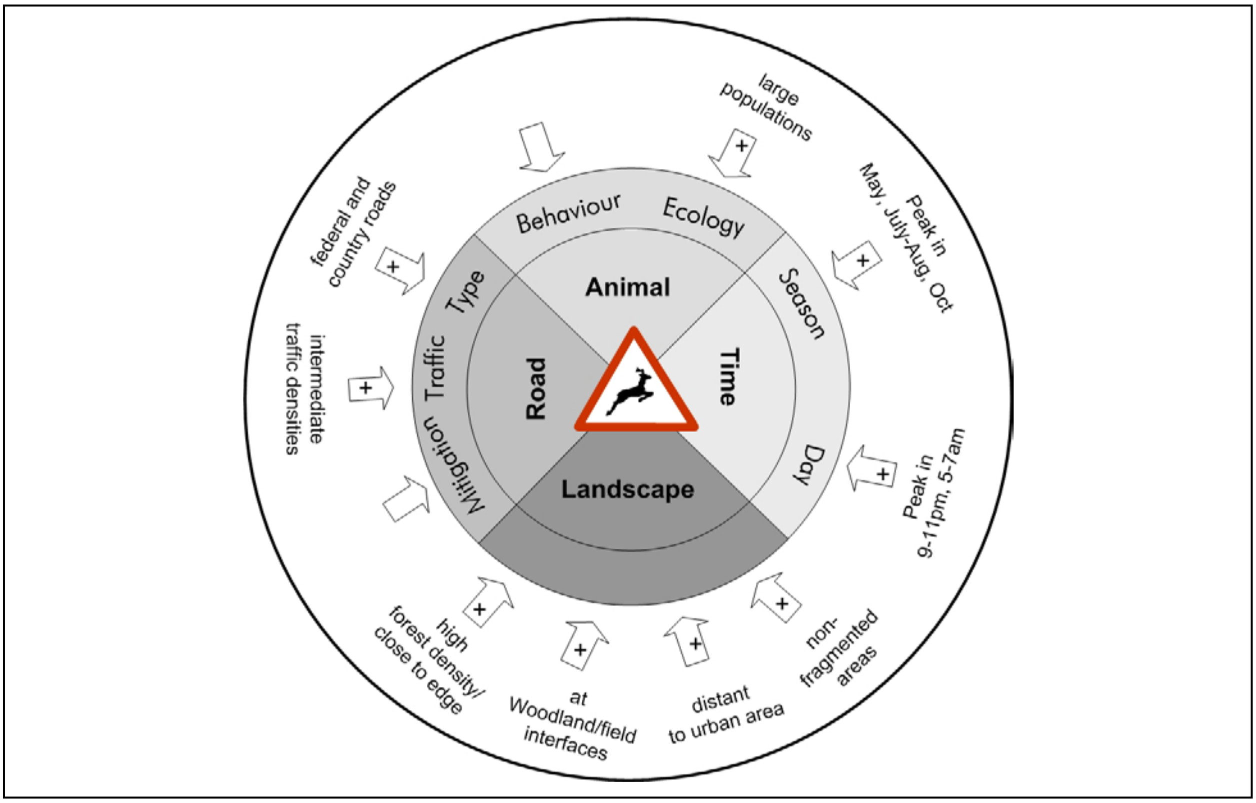

Fig. 7.1*: Factors influencing the locations of roe deer and wild boar collisions in Germany (the schema is adopted from Seiler (2003b), modified and filled up with results from our study).*

Against this background the development of mitigation strategies is crucial. Measures constructed to mitigate road kills have been applied in many places (Romin and Bissonette 1996, Putman 1997). For example, fences and chemical repellents are used to prevent road crossings, and culverts and green bridges are in use to enable safe road crossings. Also, attempts are made to alter traffic patterns by reduced speed limits or by temporary road closing (Seiler 2003b). It is imperative to apply such measures at the right place. When being constructed at the wrong location, mitigation measures are not well accepted by animals. Such non-effective investments are not economically lucrative and increase the dissatisfaction of decision makers in science. Studies figuring out where accidents occur are an essential contribution of science to decision making, because based on such findings we can infer recommendations where to build up mitigation measures.

When applying our results it is imperative to pay attention to the scale. Our analysis was not aimed at describing local-scale characteristics at accident sites. We rather aimed at analysing the effects of landscape structure and pattern on collision hotspots. Local parameters, e.g. wildlife warning signs or inline visibility on roads, probably influencing the occurrence of accidents have not been included in our analysis, because such information is neither available in remotely sensed data, nor can it be collected in the field at large scales. Therefore our results should not be used to determine the explicit location of wildlife crossings, but for the development of landscape-scale concepts for the de-fragmentation of the existing road network.

7.6 CONCLUSIONS AND IMPLICATIONS

The overall concept of our modern society is a sustainable development. The discussion on sustainability is affected by conflicts, especially in the field of road construction and mobility. The ecological perspective is opposed to the desire for increased networking and globalization. The increase in road networks is a symbol for social values and seems to rank first in the political Agenda. Hence, it seems barely justifiable to equal interests of species and habitats with economic and social needs, as envisioned by the ideal of a sustainable society.

At present however, it becomes apparent that not only species and habitats, but also society has to pay the ´toll of the automobile´ (see Stoner 1925). The German Council of Environmental Advisors states that the consequential damage of road traffic on environment and human health is inadmissibly high (SRU 2005). Problems are related to four subject areas next to nature and landscapes: traffic safety, pollution, traffic noise, quality of life and climate. Participation in road traffic is still one of the most dangerous activities in everyday life. Plumb, particle, nitrogen oxide and ozone pollution are a threat to human health. There is an increasing risk for lung cancer caused by sooty particles of diesel engines, and the development of chronic respiratory problems. The percentage of people feeling pestered by traffic noise is high with 60% of the German population. 15.6% of the German population are exposed to daily traffic noise levels above 65 dB(A), increasing the risk for cardiovascular diseases, and sleep disturbances affecting vitality. Carbon dioxide exhaust quintupled since 1960, and though a reduction of exhaust per driven kilometre was achieved due to technical developments, this success is levelled by a disproportional increase in kilometres travelled (SRU 2005).

Impacts of roads and traffic on environment and human health seem to be inadmissibly high at present. One basic reason is that ´traffic´ was the sole object of mobility policy in the past decades. Politicians intended to meet peoples´ requirements in mobility by constructing an appropriate traffic infrastructure. However, in contrast they should aim at developing a low-risk, environmentally sustainable locomotion, which does not necessarily have to restrict mobility.

The Council of Environmental Advisors in Germany (SRU 2005) suggests an agreement on concrete environmental standards and quality objectives as a global strategy towards sustainable mobility. With regard to traffic safety, pollution and climate this means a reduction of road casualty rates and immission to maximum permissible values. With regard to landscape and wildlife, this means an introduction and development of quantitative thresholds limiting further increases in landscape fragmentation.

When developing strategies for the establishment of environmental sustainable mobility, an integrative approach seems to be beneficial. For example, Gerlach (1995) has shown that a deconstruction and renaturation of existing roads is possible without disprofit for traffic function, but with an enormous gain in natural area. Mitigating wildlife accidents in traffic is beneficial not only for wildlife but also for traffic safety. Combining species conservation and traffic safety towards an integrative approach will merge funds and effort consensus on both sides, since causers take an active part in troubleshooting. In Switzerland, government works on a restoration of the current highway network for wildlife purposes (Righetti 1997). The densely populated Netherlands developed methods for a sustainable rural road network by means of traffic calming (Jaarsma 1997, Jaarsma and Willlems 2002), and already coupled de-fragmentation to the political Agenda (Canters and Cuperus 1997, van Bohemen 1998). The Dutch program aims at identifying problem spots, developing solutions, and prioritising actions. Until 2010 all priority de-fragmentation hotspots should be addressed by constructing mitigation measures (van der Grift 2005), and costs will (surprisingly) be paid by the ministry of transport.

In Germany, there has been little public awareness and/or political acceptance for such programs thus far. Possibly because the German road network (consisting of 230,800km) is not yet "dense enough" to internalize the severity of the problem. Certainly, there was not enough research on road effects in Germany, and too much research focused on local-scale road effects underestimating the full extent of the environmental problem. Against this background, the thesis in hand is of high pratical relevance, as it contributes to the rare knowledge about landscape-scale effects of roads on wildlife. Furthermore, the issues raised have straightforward implications for scientific funding organisations, planners and decision-makers.

First, together with seven road ecologists I have identified feasible study designs of reasonably high inferential strength analysing local-scale road effects (chapter 2). A funding agency will maximize the scientific value and cost effectiveness of research by giving high priority to these studies, because an investment in a good experiment is actually more cost effective than a series of "shot-in-the-dark" attempts.

Second, I demonstrated that, for landscape-scale issues, strong weight of evidence is unattainable in practice (chapter 2 and 4). For such questions, decision makers must not demand better proof before incorporating scientific results into the planning process. Seeing that, decision makers must develop general normative decision-making principles and approaches for judgment under uncertainty.

Third, state wide documentations about the status and historical development of landscape fragmentation are an important tool to uncover a persistent environmental problem (chapter 3). Quantification enables comparison of countries, detects thresholds, and supports public awareness. I have produced a basis for such a monitoring in Hesse, and recommend using m_{eff} and standardized criteria for barrier elements for a regular update.

Fourth, when analysing the effects of road networks on wildlife populations the inferential strength of the studies undertaken increases with the quality of wildlife data available (chapter 4 and 5). Hence, I urgently recommend conducting large scale monitoring programs, and improving data quality to accomplish a basis for further analysis.

Fifth, I demonstrated that road networks affect population abundance at the landscape-scale (chapter 4 and 5). Against this background, I recommend enlarging environmental impact assessment studies (EIA/UVP) for proposed roads to a regional scale and beyond. Thus far, only single road planning projects or even single road sections are considered (Reck 1993, Schupp 2005). Large-scale relationships such as traditional migrations of large mammals are not considered in these sectoral planning approaches (Georgii et al. 2002). I recommend evaluating planned road projects in the context of the pre-existing road network, large-scale faunistic processes and interactions, because local-scale evaluations obviously underestimate the extent of ecological road effects.

Sixth, I demonstrated that road networks increase wildlife mortality on roads (chapter 4), and decrease wildlife populations (chapter 4 and 5). This underlines the account of planners and road construction offices to limit a further increase of road networks, given that the protection of wildlife species is an overall aim of our society.

Seventh, I showed that science is able to provide models predicting locations of vehicle-wildlife accidents (chapter 6). I recommend using these models for bundling mitigation efforts in the context of a German de-fragmentation program.

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ERKLÄRUNG

Ich erkläre: Ich habe die vorgelegte Dissertation selbständig und ohne unerlaubte fremde Hilfe und nur mit den Hilfen angefertigt, die ich in der Dissertation angegeben habe. Alle Textstellen, die wörtlich oder sinngemäß aus veröffentlichten Schriften entnommen sind, und alle Angaben, die auf mündlichen Auskünften beruhen, sind als solche kenntlich gemacht. Bei den von mir durchgeführten und in der Dissertation erwähnten Untersuchungen habe ich die Grundsätze guter wissenschaftlicher Praxis, wie sie in der „Satzung der Justus-Liebig-Universität Gießen zur Sicherung guter wissenschaftlicher Praxis“ niedergelegt sind, eingehalten.