It causes more isolation between populations. But also a high mortality causes less exchange and more isolation which illustrates that mortality and barrier effect are linked together and are road and species specific. This can be understood because big mammals (e.g. deer) will cross all kinds of roads and even motorways much easier than smaller species (e.g. mice) indicating a less strong barrier effect. But the big species have a higher risk to die during crossings than the smaller ones (high mortality effect). Both kinds of effects are ilustrated in Figure 1.

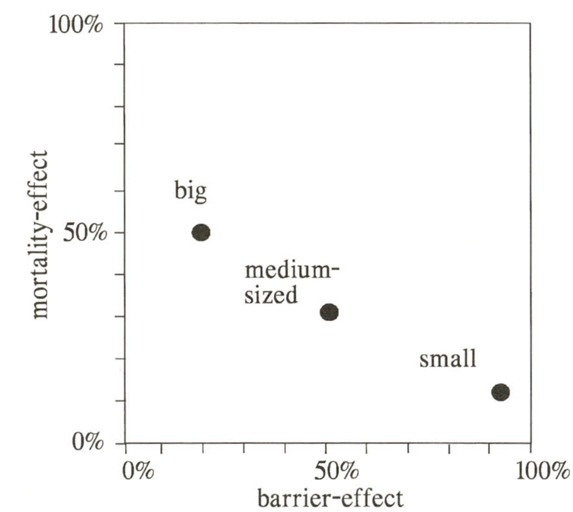

Figure 1.
The barrier and the mortality effect for a big, a medium sized and a small mammal species.
The effects are given in a relative way for a mean four lane motorway without mitigating measures (Verboom 1994).

Possible effects: some examples from the Dutch field

As is said before several kinds of effects of infrastructure can be expected on the level of individuals and populations. But we have to be sure that these effects really take place and that they can be measured before we can analyse, forecast and evaluate them.

Population level
On the population level a disturbance effect on density caused by noise from humans and cars has been described for birds (Reijnen 1995) and mammals (Fletcher & Busnel 1978). To find out if such an effect also exists for a medium-sized species like the red squirrel (Sciurus vulgaris) research was started on the density of squirrel nests. Individual squirrels of both sexes live solitary and use several nests (dreys). So nests are a good relative measure for squirrel abundance. The species prefers coniferous woods as an optimal habitat. So, on twenty locations along highways, that differ in traffic density, reference and effect plots of five ha each were selected in coniferous woods. On each location an effect plot was along and a refence plot within 500 m from the highway. Within each pair of plots other (possible disturbing) factors were kept constant. So plots only differed in noise load. The locations were situated in different parts of The Netherlands.
A statistically significant effect of the noise load on drey density could not

be identified. So an effect of traffic noise on squirrel density seems not plausible. But because of the high variation in nest density in both types of plots and the relatively small sample size this result holds only when the nest density of effect and reference plots differ more than 50%.
For smaller differences and for densities in other types of woodland we can not draw the same conclusion.

Individual level
Effects of infrastructure on the individual level are known for the badger based on analyses of traffic victims (Broekhuizen et al. 1994). Of this species annually approximately 300 traffic casualties are recorded but a number up to 400-450 is assumed to be more realistic. The high number represents 20% of the estimated summer population of 2,200 animals (Wiertz 1993). Victims are mainly adults of which 44% represents reproductive females. The high percentage of reproductive females means that yearly around 170 cubs which are still dependent on their mothers also die. This represents about 10% of all litters. When the adult mortality and the loss of litters because of car accidents are compared with a natural adult mortality of approximately 15% and a juvenile mortality of approximately 50% (Lankester et al. 1991) it can be concluded that the birth/death ratio of single social groups and local populations, and hence their chance of extinction, will be influenced in a negative way. This seems to be illustrated at a few places in The Netherlands where mitigating measures, mainly tunnels, were taken and the number of badger setts increases. But we are not sure that other positive factors have not caused the increase.

Corridors
In The Netherlands some data exist on the small mammal species that can live in road verges. Between them are the common vole *Microtus arvalis* and the rabbit *Oryctolagus cuniculus*. It is also known that both species have expanded their distribution area in the western part of the country (unpublished data, Province of Zuid Holland) and that both species inhabit verges also in this part of the country suggesting that the verges played a positive role during their spread. But still the role of road verges acting as dispersal corridors has not been studied seriously.
The effects of infrastructure on area and quality of the habitat of species and the mortality and barrier effects show that all can influence the birth/death ratio of populations and the connectivity between populations. This illustrates that effects should be analysed and evaluated on the population level (for one or even a network of populations) in stead of the individual level as is mostly done. So the question now is how can this be done.

Effects and forecasting

Methods
To predict how large effects can be, we always need some description of reality through a model. Which model is used depends on the knowledge of the species and its habitat studied and model characteristics that are thought to be important. For instance, in forecasting effects a formalized quantitative prediction that can be extrapolated to other situations or species

is mostly desired. Nevertheless often only a prediction can be made using "expert-judgement". Formalization leads to so called "expert systems". Quantitative predictions are possible using mathematical models.
They include regression and multivariate models, cause-effect relationships and theoretical statistical distributions. But such analytical models do not describe the working mechanisms (black box models), so there are problems with extrapolation. This is not the case with simulation models, but they can be more complex and need good field data (Jeffers 1978).
Effect forecasting is mostly done for specific situations (locations) and effects can be analysed before (problem detection) or after (evaluation) measures are taken.

Examples
For the badger in The Netherlands problem detection and evaluation of measures are both normally done by direct interpretation of the numbers of traffic victims. This is possible because dead and wounded badgers are registrated by an excellent network of volunteers, coordinated by the NGO "Vereniging Das en Boom". Numbers found are used to locate places where measures (tunnels, fences) should be taken. If this is done a drop in the number of victims is interpreted as that the measures are effective.
Because of the high numbers of casualties in some regions a quick and systematic analysis method is needed. For this reason we have developed a statistical method (based on kernel density estimation) to estimate densities of traffic casualties in relation to the surrounding landscape. The required information consists of GIS based-data of victims and highways or roads (Apeldoorn et al. 1995). It is evident that with this method nothing is said about effects on the population level. To make this possible we have build a simulation model for the badger in which the mortality data can be used and which describes the extinction chance of (local) populations (Verboom et al. 1996, Lankester et al. 1991). It is an individual based stochastic model. This means that individuals have some probability of dying, reproducing, dispersing and so on. The model describes the behaviour of social groups (reproductive units) of badgers that live in a group territory and that exchange individuals so that the groups are combined into so called local populations. The behaviour of several local populations in a metapopulation can be studied because they are connected by means of (long distance) dispersing animals (Fig. 2).

All social groups in a local population behave similar but their number and behaviour differ between clusters. Results of simulations for a concrete situation with two local populations and a country road are given in Table 1. One population has only one group territory (local pop 2) with one adult male and one adult female badger. In the second population (local pop 1) nine adult badgers live divided over two group territories. The table illustrates the effect of adult (road)mortality on the mean number of years that both single populations and their combination are present when badgers can exchange between them by crossing the road. Calculations were done with a natural adult mortality of 15%.

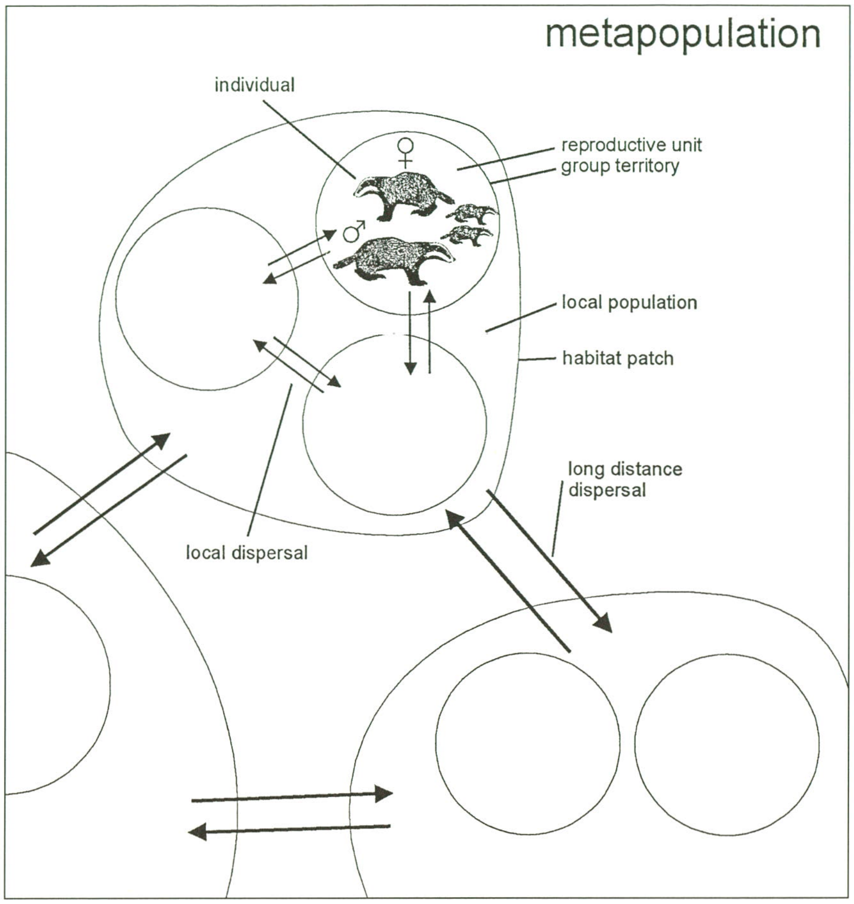

Figure 2.
The spatial structure of the badger metapopulation in the simulation model. Individuals have an age and sex and are grouped in reproductive units that inhabit group territories.
Several territories make a local population, several local populations make a metapopulation.
Reproductive units and local populations may be (temporarily) absent.

Table 1.
Persistence (years) and road mortality (m).

	m 0	m 0,1	m 0,25	m 0,4
local pop. 1	69	44	34	24
local pop. 2	21	18	24	25
local pop. 1+2	85	60	51	45

The table shows that when all badgers cross the road successfully, the population with two group territories will survive with a mean of 69 years and the second one with 21 years. When road mortality increases from 10% up to 40%, the chance of survival of the population with two groups of badgers drops to 24 year. The second population shows a little increase of its mean persistence up to 25 years. This small increase can be explained because most migrants came from the biggest population and if they cross successfully they reproduce in the small one. A road mortality of 40% can also be interpreted as the combination of a natural adult mortality of 15% and a road mortality of 25%. For the conrete situation with the three group territories this means that their survival decreases with nearly 50% when a natural mortality is combined with two to three dead badgers per year.
The table illustrates how this model can be used to forecast the mortality

effect. Also the barrier effect can be analysed by means of the model and it can be used to evaluate the effect of mitigating measures like tunnels and fences and compensation measures. A second version of the model simulates the same population dynamics but the dispersal process is made spatially explicit because it is described using GIS-based landscapes (Schippers et al. 1996). The presented results show that we do not only have to count fragmented mammals but we also have to know what they mean for fragmented populations.

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Floristic impoverishment by changing unimproved roads into metalled roads

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Abstract

A dataset of 904 vegetation relevés and 717 plant species in road verges in Flanders (Belgium) was tested statistically for species associated with unimproved or metalled (asphalted, macadamized) roads. 44 species occur only along unimproved roads. 124 are associated significantly more with unimproved roads. Many of the species of these groups are interesting species for nature conservation. 94 species occur significantly more along metalled roads. This group is of no importance for nature conservation. Differences between the effects of asphalted or macadamized roads are hardly detectable. This leads us to suppose that indirect effects of the metalling are more important than direct (chemical) effects. 19 of the 37 vegetation types are better developed along unimproved than along metalled roads. If improving roads decreases quality of flora and vegetation, it is clear that this means a further habitat fragmentation by removing the corridor effect of road verges.

Introduction

The knowledge about the flora and vegetation of unimproved roads in Flanders is very limited. Publications of the last decades mainly deal with sunken roads and church paths (Abts 1972, Meynen 1978, 1985, Cools 1979, Geldhof 1979, Moons & Roosen 1984, Robijns 1984, Thewissen 1985, Stevens 1987, 1988). Berten (1987), Crijns (1987) and Dupae & Vanhaeren (1987) also deal with the flora and vegetation. None of the papers however considers the actual differences between flora and vegetation of unimproved and metalled roads. Stevens (1988) states that no data are available to evaluate the effects of macadamized roads from a scientific point of view. In the Netherlands Westhoff (1967) and Anonymous (1970) pay attention to the flora and vegetation of unimproved roads. The former deals with some particular species of unimproved roads of importance for nature conservation. The latter mentions a harmful effect of present road metalling for biological values, but does not consider particular species or vegetations. Their arguments to protect unimproved roads appeal to recreational, scientific and (art)historical values as well as to values of the landscape.

Our own observations during the inventory of road verges and the mentioned literature led us to this investigation. Several hypotheses were made before testing statistically the occurrence of particular species. The influence of unimproved roads on the vegetation is often positive in the case of sunken roads or in the case of a trampling gradient (Westhoff 1967). Metalled roads however have many negative influences: enrichment by nutrients, calcium and the dehydration from accompanying water controlling activities (wearing away the verges, deepening the dikes) seem to be very important (cf. Tamm & Troedsson 1955). While the flora and vegetation of unimproved roads still often resembles that of the surrounding countryside (Sykora et al. 1993), they change drastically when the road is metalled. In many places the road verges of unimproved roads are refuges for plants, which previously occurred in the surrounding countryside. Metalling these roads is extra pernicious, because it leads to the disappearance of the original flora and vegetation of the entire site.

Verkaar (1988, 1990) considers road verges as ecological corridors that can minimise the effects of isolation in an otherwise fragmented landscape.

If however improving roads decreases quality of flora and vegetation, it is clear that the improving means a further fragmentation of the landscape by removing the corridor effect.

In this paper we try to give an idea about the statistical differences in flora and vegetation of unimproved and metalled roads.

Methods

Sampling

Between 1986 and 1989, 904 vegetation relevés (20 m^2) were made along Flemish road verges, in 113 different municipalities. The vegetation relevés were carried out on the basis of the Braun-Blanquet (1964) method. The cover estimation was done in the decimal scale for relevés of permanent quadrats (Londo 1976). Sampling took place along metalled as well as unimproved roads. The criteria for sampling were species richness, rareness of species, completeness of the vegetation type, representative sampling of all soil types as well as aesthetical value of the verges. 232 verges were sampled along macadamized roads, 400 along asphalted roads, 10 along paved roads and 262 along unimproved roads. "Unimproved roads" were originating only from trampling on the original soil as well as roads with a certain amount of broken stones.

Nomenclature follows that of De Langhe et al. (1988).

Analysis

The dataset recorded which plants and which vegetation types were associated with unimproved roads. Firstly, the species occurring only along unimproved roads were looked at. Then tests were carried out to see which species were significantly more (P = 0.01, P = 0.02 and P = 0.05) associated with unimproved or metalled roads. The statistical package SPSS (Norusis 1990) was used. Firstly the cover of the species in the relevés was transformed into presence/absence. The nonparametric procedure "chi square" choosed itself if the Fisher exact probability test or the Chi square test for two independent samples was more appropriate, depending on the sample size in a 2 x 2 contingency table (metalled 0 or 1, species present 0 or 1). A two-tailed test was used. For the "metalling avoiders" it was

investigated whether there was a difference between asphalted of macadamized roads, with the same tests.
Ellenberg et al. (1991) indicator numbers were used as descriptive parameters of the obtained species lists to characterize them. Percentages of the different classes were calculated for each group.
From the indicator values of Ellenberg et al. (1991) a characteristic indicator value (quantitative mean) for the 904 relevés was calculated. The quantitative means of relevés from verges along unimproved roads were compared with the ones of verges along metalled roads by the same chi square procedure as described already above. The preference of vegetation types for unimproved roads was examined by looking at the degree of threat of the component species.
Finally some statistical data were computed with the computer program SPSS (Norusis 1990) to explain the observations. A literature analysis and original calculations provided the results about unimproved road length, width etc.

Results

Official statistics about the length and the surface of unimproved roads in Flanders are very scarce, because most unimproved roads are private or municipal property. Some local studies however indicate trends.
In Assebroek (Brugge, Province of West-Flanders), a municipality in the Sandy Region, where unimproved roads still occur frequently, 45% of the roads were still unmetalled untill 1962. In 1995 only 22% remain.
This means an annual decline of 0.7%. By comparison in the Netherlands the length of unimproved roads declined 4.2% yearly between 1966 and 1968 (Anonymous 1970). The municipality of Torhout (Province of West-Flanders), another municipality in the Sandy Region, succeeded in protecting 18 km of church paths, in 1994. In Wenduine, a municipality in the dunes, no unimproved roads were left by 1962. Stevens (1987, 1988) mentions 92 km of sunken, unimproved roads for the Province of Limburg.
The mean width in a sample survey was 2.7 m compared to 3.9 for asphalted and 4.2 m for macadamized roads. It seems logical that the metalling is at the cost of the width of the road verge and therefore threatens the flora and vegetation. From our own measurements however no correlation between unimproved roads and wider verges could be found. The mean verge width along unimproved roads is only 3.0 m (sd = 2.8) compared with 3.9 m (sd = 2.8) along metalled roads.

44 species (about 5% of the verge flora) occur only along unimproved roads (Table 1).124 species (17%) occur significantly more along unimproved roads (Table 2). A difference between the negative influence of macadamized or asphalted roads is hardly detectable. Only a very limited number of species occurs significantly less along particular macadamized or asphalted roads. *Dryopteris filix-mas* ($P = 0.01$), *Holcus mollis*, *Origanum vulgare* and *Vicia sepium* ($P = 0.05$) avoid macadamized roads. *Stachys palustre* is the only species occurring significantly less often ($P = 0.05$) along asphalted roads than along unimproved or macadamized roads.
94 species (13%) occur significantly more along metalled roads (Table 3).

Table 1.
Species occurring only along unimproved roads.

1 = occurrence in relevés, 2 = indicator value for nitrogen supply (Ellenberg et al. 1991), 3 = indicator value for soil acidity (Ellenberg et al. 1991), 4 = indicator value for soil humidity (Ellenberg et al. 1991), species in bold: red list species

	1	2	3	4
Aneura pinguis	1	?	?	?
Aulacomnium androgynum	3	?	?	?
Brachypodium pinnatum	1	4	7	4
Briza media	1	2	x	x
Bromus erectus	1	3	8	3
Caltha palustris	4	x	x	8
Campanula trachelium	3	8	8	5
Campylopus introflexus	2	?	?	?
Carex echinata	1	2	3	8
Carex spicata	1	?	?	?
Centaurium pulchellum	2	x	9	7
Cicendia filiformis	1	?	3	9
Conocephalum conicum	1	?	?	?
Conopodium majus	1	?	?	?
Corydalis claviculata	5	?	3	5
Drosera intermedia	5	2	2	9
Equisetum sylvaticum	1	4	3	7
Fossombronia sp.	1	?	?	?
Galeopsis bifida	1	?	?	?
Gentiana pneumonanthe	2	2	x	7
Geranium columbinum	1	?	?	?
Geranium phaeum	2	?	?	?
Hyacinthoides non-scripta	4	6	7	5
Illecebrum verticillatum	6	2	2	7
Inula conyzae	3	3	7	4
Jungermannia gracillima	3	?	?	?
Lamium amplexicaule	2	7	7	4
Legousia speculum-veneris	1	3	8	4
Leonurus cardiaca	1	9	8	5
Leucobryum glaucum	1	?	?	?
Linum catharticum	1	1	x	x
Maianthemum bifolium	2	3	3	x
Melica uniflora	1	x	6	5
Orchis purpurea	1	x	8	4
Pentaglottis sempervirens	1	?	?	?
Phyteuma nigrum	1	4	5	6
Platanthera bifolia	1	x	7	5
Sambucus ebulus	1	7	8	5
Sanguisorba officinalis	1	3	x	7
Sanicula europaea	1	7	8	5
Sherardia arvensis	1	5	8	5
Sphagnum auriculatum	1	?	?	?
Sphagnum tenellum	1	?	?	?
Stellaria palustris	1	2	4	8

19 of the 37 vegetation types (51%) distinguished in Flemish roadsides (Zwaenepoel 1993), are better developed along unimproved than along metalled roads (Table 4). This mainly means that many rare species occur in those vegetation types whereas they are lacking in the same vegetation types along metalled roads. This is confirmed by calculating the relevé mean (arithmetic mean for the Atlantic part of Belgium) of the rarity indications for species of Stieperaere & Fransen (1982). This mean indicates significantly (Chi square test, P = 0.01) more rare plants along unimproved roads (relevé mean 7.3 versus 7.8). It does not mean that the relevés of verges along unimproved roads are richer in species. The evidence is other-

wise. On 20 m^2 stretches on average 26.9 species (sd = 8.4) occur along unimproved roads while 29.5 species (sd = 8.9) are found along metalled roads.

Table 2.
Species occuring significantly more along unimproved roads (Chi square test).

Species in bold: red list species

P = 0.01

Agrostis vinealis
Ajuga reptans
Anemone nemorosa
Anthoxanthum odoratum
Atrichum undulatum
Aulacomnium androgynum
Betula pubescens
Brachypodium sylvaticum
Calamintha clinopodium
Calliergonella platycarpa
Caltha palustris
Campanula rotundifolia
Campanula trachelium
Cardamine pratensis
Carex acuta
Carex pilulifera
Dactylorhiza maculata
Dicranella heteromalla
Dicranum scoparium
Drosera rotundifolia
Dryopteris carthusiana
Dryopteris dilatata
Epilobium angustifolium
Equisetum palustre
Erica tetralix
Eurhynchium hians
Eurhynchium praelongum
Festuca gigantea
Filipendula ulmaria
Fragaria vesca
Galeopsis tetrahit
Geum urbanum
Hedera helix
Hieracium pilosella
Humulus lupulus
Hypnum cupressiforme
Illecebrum verticillatum
Inula conyzae
Iris pseudacorus
Isopterygium elegans
Juncus acutiflorus
Juncus effusus
Knautia arvensis
Lonicera periclymenum
Lophocolea bidentata
Lophocolea hetrophylla
Luzula multiflora
Luzula congesta
Lysimachia nummularia
Mentha arvensis
Mercurialis perennis
Mnium hornum
Moehringia trinervia
Molinia caerulea
Montia verna
Myosurus minimus
Nardus stricta
Pellia epiphylla
Plagiomnium affine
Poa nemoralis
Polytrichum commune
Polytrichum formosum
Polytrichum piliferum
Potentilla erecta
Primula vulgaris
Prunus serotina
Ranunculus ficaria
Rhinanthus angustifolius
Ribes uva-crispa
Rubus fruticosus
Salix repens
Scleropodium purum
Sieglingia decumbens
Solidago virgaurea
Sorbus aucuparia
Stachys sylvatica
Stellaria graminea
Teucrium scorodonia
Valeriana officinalis
Veronica chamaedrys
Viola reichenbachiana

P = 0.02

Angelica sylvestris
Arum maculatum
Centaurium pulchellum
Frangula alnus
Geranium robertianum
Polygonatum multiflorum
Veronica serpyllifolia

P = 0.05

Adoxa moschatellina
Agrimonia eupatoria
Blechnum spicant
Calluna vulgaris
Carex acutiformis
Carex hirta
Carex remota
Chaerophyllum temulum
Dipsacus sylvestris
Epilobium hirsutum
Epilobium palustre
Gentiana pneumonanthe
Geranium phaeum
Hordeum secalinum
Hydrocotyle vulgaris
Luzula sylvatica
Lysimachia vulgaris
Melampyrum pratense
Mycelis muralis
Myosotis cespitosa
Myosotis scorpioides
Pedicularis sylvatica
Pleurozium schreberi
Polygala serpyllifolia
Populus tremula
Primula elatior
Quercus robur
Rorippa amphibia
Sphagnum compactum
Stellaria holostea

Table 3.
Species occuring significantly more along metalled roads (Chi square test).

P = 0.01

Achillea millefolium
Agrostis capillaris
Anthriscus caucalis
Anthriscus sylvestris
Arrhenatherum elatius
Artemisia vulgaris
Barbula convoluta
Brassica nigra
Bromus hordeaceus
Bromus sterilis
Bromus tectorum
Capsella bursa-pastoris
Carduus crispus
Cerastium glomeratum
Cerastium semidecandrum
Cirsium arvense
Convolvulus arvense
Conyza canadensis
Crepis capillaris
Dactylis glomerata
Daucus carota
Diplotaxis tenuifolia
Elymus athericus
Elymus repens
Equisetum arvense
Erodium cicutarium
Erophila verna
Festuca rubra
Geranium molle
Geranium pusillum
Hordeum murinum
Lamium album
Lolium perenne
Medicago lupulina
Melandrium album
Pastinaca sativa
Phleum arenarium
Phragmites australis
Plantago lanceolata
Plantago major
Poa pratensis
Polygonum convolvulus
Potentilla anserina
Potentilla reptans
Rubus caesius
Sedum acre
Senecio vulgaris
Sisymbrium officinale
Sonchus asper
Stellaria media
Tanacetum vulgare
Taraxacum vulgare
Tortula ruralis
Tragopogon pratensis
Trifolium campestre
Trifolium dubium
Trifolium pratense
Trifolium repens
Veronica arvensis
Vicia sativa

P = 0.02

Alopecurus pratensis
Cerastium fontanum
Melilotus alba
Plantago coronopus
Pulicaria dysenterica
Senecio jacobaea
Vicia hirsuta

P = 0.05

Aethusa cynapium
Allium vineale
Arabidopsis thaliana
Arenaria serpyllifolia
Barbula unguiculata
Bellis perennis
Brachythecium albicans
Bryonia dioica
Cardaria draba
Cerastium tomentosum
Chenopodium polyspermum
Geranium dissectum
Heracleum sphondylium
Hippophae rhamnoides
Leontodon saxatilis
Matricaria inodora
Matricaria recutita
Oenothera biennis
Papaver dubium
Ranunculus flammula
Raphanes raphanistrum
Reseda lutea
Sonchus arvensis
Sonchus oleraceus
Stellaria pallida
Vicia lathyroides

Discussion

The species occurring only along unimproved roads are mainly species of soils poor in nutrients, which means that their indicator value for nitrogen supply (Ellenberg et al. 1991) does not exceed 4 (63%), of acid soils (39%), of soils poor in nutrients and of wet soils (34%), woodland species (32%), lime indicative species (31%), species presenting an optimum in a trampling gradient (21%) or heathland species (14%). From the indicator values for nitrogen supply (Ellenberg et al. 1991) a characteristic indicator value

Table 4.
Vegetation types which are better developed under unimproved road circumstances.

For nomenclature: cf. Zwaenepoel (1993).
1 = with exclusive species of unimproved roads
2 = threatened by enrichment by calcium of metalled roads
3 = sunken roads important for the vegetation type
4 = trampling gradient of unimproved roads important
5 = undamaged landscape important (unspoiled dike, ± natural adjacent wood, undisturbed traditional landscape) important,
6 = gradient in soil texture (clay-sand, + or - calcium, + or - silt) important.

	1	2	3	4	5	6
Cardamine pratensis-Ranunculus ficaria				x		
Stellaria holostea-Atrichum undulatum	x					
Aegopodium podagraria-Rumex obtusifolius	x		x			
Geum urbanum-Poa nemoralis	x				x	
Galium mollugo-Plantago media	x		x			x
Hieracium umbellatum-Linaria vulgaris		x				
Lotus uliginosus-Ranunculus flammula		x	x			
Anthriscus caucalis-Arrhenatherum elatius				x		x
Origanum vulgare-Vicia tetrasperma	x		x			x
Stellaria media-Capsella bursa-pastoris		x			x	x
Poa annua-Plantago major	x	x		x		
Juncus tenuis-Veronica serpyllifolia	x			x	x	
Teucrium scorodonia-Dicranella heteromalla	x	x			x	
Juncus conglomeratus-Peucedanum palustre		x				
Galium saxatile-Veronica officinalis		x			x	
Hypochoeris radicata-Rumex acetosella		x				
Polytrichum piliferum-Aira caryophyllea			x	x		x
Erica tetralix-Molinia caerulea	x	x		x	x	
Hypericum humifusum-Juncus articulatus	x	x		x	x	

(quantitative mean) for the relevés was calculated. The mean along unimproved roads (5.1, sd = 1.7) is significantly lower (Chi square test, P = 0.01) than the mean along metalled roads (5.5, sd = 1.4). The characteristic indicator value for soil acidity (5.1, sd = 1.6) also is significantly lower (P = 0.01) than along metalled roads (5.6, sd = 1.4). Also the median of the measured pH/H_2O (6.6, sd = 0.8) is significantly lower (P = 0.01) than along metalled roads (6.7, sd = 0.8). Finally, the characteristic indicator value for soil moisture (5.7) indicates significantly (P = 0.01) wetter soils along unimproved roads.
Most of the species occurring only along unimproved roads occur also only a few times in the dataset (maximum = 6). In the statistical tests most of them are not significantly associated with unimproved roads because they are to rare to perform a Fisher exact probability test.

Nineteen of them appear on the red list (Cosyns et al. 1993). *Cicendia filiformis*, *Legousia speculum-veneris* and *Linum catharticum* are very strongly threatened; *Conopodium majus* and *Platanthera bifolia* are strongly threatened; *Briza media, Equisetum sylvaticum, Illecebrum verticillatum, Leonurus cardiaca, Orchis purpurea, Phyteuma nigrum, Sanguisorba officinalis* and *Sherardia arvensis* are threatened; *Brachypodium pinnatum, Bromus erectus, Gentiana pneumonanthe, Inula conyzae, Melica uniflora* and *Sanicula europaea* could become threatened in the future.

Ecologically specialized and therefore for nature conservation particular interesting species such as *Platanthera bifolia*, *Linum catharticum* and

Cicendia filiformis are already mentioned by Westhoff (1967) as associated with, or presenting their optimum in the gradient of soil compaction, due to differentiation in trampling along paths, tracks or roadways. Before this time the beneficial effect of pedestrian and vehicular traffic on certain plants was only supposed to be limited to a number of nitrophilous species of compact soil, *e.g. Plantago major, Poa annua* and *Polygonum aviculare.*

The spectrum of species occurring significantly more along unimproved roads is resembling well that of the species occurring only along unimproved roads. Some of them already appeared in the first list. 42% are characteristic of soils poor in nutrients, 41% are woodland species, 39% occur on acid soil, 21% are heathland species, 14% are species of soils poor in nutrients and of wet soils, 9% are species which present an optimum in a trampling gradient and 8% are lime indicative species.

Eight of them appear on the red list. *Dactylorhiza maculata, Illecebrum verticillatum* and *Pedicularis sylvatica* are threatened. *Calamintha clinopodium, Gentiana pneumonanthe, Hordeum secalinum, Inula conyzae, Mycelis muralis, Polygala serpyllifolia, Primula vulgaris* and *Rhinanthus angustifolius* could become threatened in the future.

The species occurring significantly more along metalled roads are mainly species of habitats rich in nutrients, which means that their Ellenberg et al. (1991) value for nitrogen supply exceeds 4 (62%), pioneers (54%) and, at a first look rather surprising, also dune species (15%). The reason is however that unimproved roads in the Flemish dunes nearly do not exist any more. 42% of the unimproved roads were recorded on sand with a lime deficiency, 24% on silty sand, 16% on silt, 9% on loamy silt, only 5% on clay, 4% on peat and less than 1% on dunesand. This explains that all dune species occurring rather frequently are indicated as associated with metalled roads. A certain number of the species in this list (*Achillea millefolium, Agrostis capillaris, Arrhenatherum elatius, Dactylis glomerata, Elymus repens, Festuca rubra, Lamium album, ...*) appear here because they are very common in road verges and so a small predominance in the category metalled roads is already sufficient to catalogue them as typical for metalled roads. As to established strategy (sensu Grime et al. 1988) mainly intermediate strategies between stress tolerators, competitors and ruderals occur (SR, CR/CSR, CR, R/SR, R/CSR, CSR, R/CR), with a weak accent to the ruderals. The higher number of ruderals undoubtedly is due to the higher degree of disturbance along metalled roads and probably the reason for the higher mean species richness. Westhoff (1967), Heindl & Ullmann (1991) and Heindl (1992) already mentioned a higher species richness in road verges as a consequence of disturbance. This does not mean that species richness in road verges is only due to disturbance. The significantly highest species richness is due to grazing under the barbed wire (Zwaenepoel 1993).

The fact that hardly any different effects can be found between macadamized and asphalted roads is in favour of the view that the chemical effects of the metalling are less important than other, indirect effects. The enrichment by nutrients and lime (of both substrates) however remain important chemical effects, given the switch of "poor" and "acid" to other species. If this switch is a direct consequence of the metalling rather than an indirect effect (due to road dust arrived with the traffic) remains an unanswered question.

Apart from the direct effects of the metalling itself, different indirect effects are responsible for the differences in the established flora. The established management is a first clear example. "Doing nothing", grazing under the barbed wire and mowing once a year with removal of the cuttings occur in verges along unimproved roads in respectively 44%, 15% and 9% of cases compared with respectively 21%, 4% and 4% along metalled roads. Mowing twice or once a year, without removal of the cuttings, mowing twice a year with removal of the cuttings and a lawn management occur along metalled roads in respectively 26%, 11%, 6% and 6% of cases, while only in respectively 3%, 6%, 1% and 1% of unimproved roads.
A second indirect reason for the differences is obviously the relation between the road-side verges and the hinterland. Unimproved roads more often cross relatively undisturbed areas with more nature and less negative impact from agriculture. The situation might be reflected in the botanical quality of the verges.

One species in the list of species associated with metalled roads is a surprise: *Ranunculus flammula* should better fit in the list of species associated with unimproved roads.

Table. 4 lists some reasons why the vegetation type is threatened by metalling the road. From this it follows that the Erica tetralix-Molinia caerulea and the Hypericum humifusum-Juncus articulatus-type are threatened most by the metalling. The large number of threatened vegetation types also explains why at the (subjective) sampling so many unimproved roads are included. It is clear that unimproved roads do not represent a third of the road verges in Flanders, but the nicer vegetations very often are to be found here.

From the lists of threatened species and vegetation types it follows that metalling the roads causes the biggest dammage in the Campine region (acid, nutrient poor), as well as in the Maas-region (calcareous).

Legal protection of the vegetation of unimproved roads is possible by the law on environmental town and country planning (March 29th 1962).
The law of August 7th, 1931 allows the possibility of classifying unimproved roads as a protected landscape (Geusens 1987). After land consolidation private roads, with 1 m of the verge at each side of the road, become municipal property. In sunken roads the entire verge becomes municipal property. For those verges this offers the possibility to apply the road verge ministerial order of the Flemish executive (June 27th 1984) and tree-felling orders (Stevens 1988). Conservation of important hinterland biotopes is the most important exterior management need. It is possible, by the order of the Flemish executive concerning vegetation alteration (December 4th 1991). The conservation of church paths is more difficult. Because they are

narrow they are more easily ploughed than other unimproved roads. The statute of public road only exists for the church paths mentioned in the Atlas of Local Roads (1846). Interrupted public use threatens the survival of private church paths: the right of public use can expire. Church paths not mentioned in the Atlas can get a public function, when public use already is established. If the Province agrees, and when the municipality accepts the maintenance costs, the church path can be included in the municipal roads. If a church path is not mentioned in the Atlas, nor used, an expropriation by the council can be considered to assign the church path a public use. A cheaper method exists, but depends on the goodwill of the adjoining landowners of a mutual agreement between the municipality and owner(s) to grant passage (Geldhof 1979).

Conclusions

The assumption that unimproved roads accomodate a considerable number of species which are threatened by metalling is confirmed by our results. Plants from nutrient poor, acid and wet biotopes are the bulk of the threatened species. In contrast, however, a smaller number of species of biotopes rich in nutrients, mainly pioneers (ruderals of disturbed areas) are associated with metalled roads. There are some very probable hypotheses about the origin of the negative effect of metalling, but further research must reveal if the chemical effects are direct or indirect consequences of the metalling. Protection of unimproved roads and their vegetation is possible by a wide range of legal measures .

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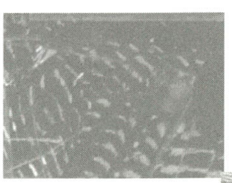
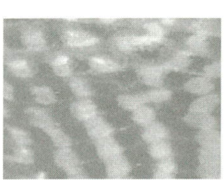
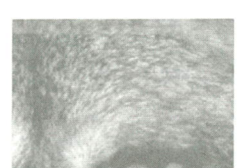
Forecasting effects and modelling







Edge effects in fragmented habitats: Implications for nature conservation

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Abstract

The importance of the use of conservation buffer zones was demonstrated by an investigation into the nature and extent of edge effects upon the plant community in habitat fragments. The impacts of a road upon adjacent heathland, and of arable agriculture upon adjacent chalk grassland were primarily eutrophication, from oxides of nitrogen in vehicle exhausts and from fertilisers. This caused an increase in the competitive ability of the more vigorous grass species such as Molinia caerulea and Bromus erectus, and a corresponding decrease in the abundance of other less competitive plant species such as Calluna vulgaris and Helianthemum nummularium. These edge effects were seen up to 200 metres into the heath, and up to 100 metres into the grassland. The relative areas of pristine habitat and that suffering edge effect were calculated. Without the use of conservation buffer zones to protect habitats from edge effects, little undisturbed habitat may be left in a fragmented landscape even though the absolute area of the fragments may be relatively large.

Introduction

Conservationists are becoming increasingly concerned about the reduction and fragmentation of habitats, which results in both direct habitat loss (Saunders et al. 1987, Soule 1986), and disturbance around the edges of the remaining habitat fragments, which may lead to progressive erosion of their value (Janzen 1986). Provided that the nature and extent of these edge effects is known, a buffer zone around the habitat can be used to absorb the edge effects and protect the core area of the fragment from change due to the surrounding land use.

The importance of conservation buffer zones to preserve fragmented habitats has been investigated by research into the edge effects arising from a variety of land uses adjacent to different semi-natural habitats in the UK Three habitats with different adjacent land-use were studied (Angold 1992):

1. the effect of a road upon adjacent dry lowland heath
2. the effect of arable agriculture upon adjacent chalk grassland
3. the effect of a housing estate upon adjacent temperate deciduous woodland

The first two studies are discussed in this paper. The field sites were:

1. an extensive area of heathland lying on either side of the A31, a major trunk road through the New Forest in Hampshire. Five research sites were established adjacent to the road, which carried 34661 vehicles

during a measured 12 hour flow in October 1990 (Hampshire County Council, U.K., pers. com. 1990).
2. a large area of open chalk grassland used for Army training operations on Salisbury Plain East, Wiltshire. Five research sites were established adjacent to three arable fields.

The edge effects generated by the road and by arable agriculture were quantified by investigations into the dispersal of pollutants, the structure and species composition of the vegetation, the growth of selected plant species, and the soil nutrient status.

The implications of the edge effect for nature conservation are demonstrated by a study of heathland vegetation in the Poole Basin, UK. The Poole Basin is an area near the New Forest which in 1759 supported a large area of heathland. This heathland was not protected from development, and the heath has been greatly fragmented. The extent of the heathlands over several centuries has been mapped (Webb & Haskins 1980, Webb 1990). These maps were used to calculate the core area of heathland remaining in the habitat fragments after taking the edge effects into account.

Methods

In the heathland, the degree of atmospheric pollution from the road was measured using nitrogen diffusion tubes to determine the flux of oxides of nitrogen. Nitrogen diffusion tubes (Robbins 1985) were left in the field for three weeks in June, 1989 and again in January 1991. Sampling stations were established at each site at 1.5, 5, 10, 25, 45, 80, 150 and 200 metres from the road, with eight replicates per distance.

The grassland sampling system consisted of 20 random samples, in each of four areas at each site: area 1 was from 0-10 metres from the field; area 2 from 15-25 m; area 3 from 40-60 m and area 4 from 80-120 m from the field edge.

In both the heathland and the grassland, the plant species composition at each sampling station was recorded within a 50 cm x 50 cm quadrat using the Domin scale of cover abundance (Mueller-Dombois & Ellenberg 1974). The performance of selected species from each habitat (*Calluna vulgaris, Molinia caerulea*, and *Cladonia impexa* from the heathland, and *Centaurea nigra* and *Bromus erectus* from the grassland) was assessed. Ten plants of each species were chosen at random from each distance at each research site during July 1991. The annual growth increment of *Calluna* and *Molinia*, and the height of *Centaurea nigra* and *Bromus erectus* was measured, and the nitrogen and phosphorus content of the shoots was assessed by auto-analysis following an acid digest (Allen et al. 1974). The diameter of each lichen clump, the height of the lichen stems, and the number of first order branches per stem were recorded from the *Cladonia* samples.

The soil nutrient levels were investigated at each of the sampling sites used in the vegetation survey. An assay of the organic content of the soil (loss on ignition) was used to estimate the carbon content, and the nitrogen and phosphorus contents were measured by autoanalysis following a sulphuric acid: hydrogen peroxide acid digest (Allen et al. 1974).

The implications of edge effects were investigated using the maps of heathland habitat present at various times (Webb & Haskins 1980).

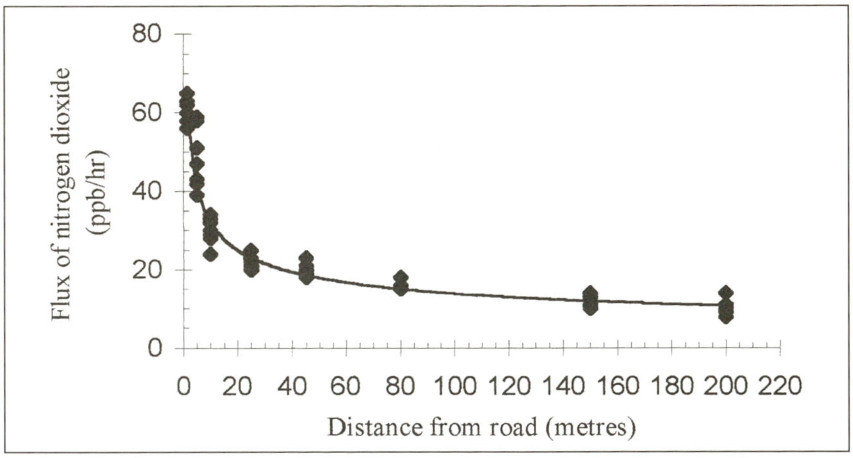
The area and perimeter of each heathland fragment extant in 1759, 1811, 1934, 1960 and 1978 were determined using a Kontron IBAS 2 image analyzing computer on the mapped outline of each fragment. The core area model developed by Laurance & Yensen (1991) was then used to calculate the core area of each fragment after subtracting the edge habitat for theoretical edge effects of 0, 25, 50, 100 and 200 metres.

Results

The impact of a road upon adjacent heathland

The atmospheric flux of nitrogen dioxide was strongly elevated near the road, to 60ppb/hr, and decreased exponentially with distance from the road at each site (Fig. 1).

Figure 1.
Change in flux of atmospheric nitrogen dioxide with distance from the road.



The cover abundance of *Calluna vulgaris* and of *Cladonia impexa*, and the mean number of species of lichen per quadrat, tended to decrease near the road, whereas the cover abundance of *Molinia caerulea* tended to increase. These trends were subtle and not statistically significant, so a more detailed analysis of the composition of the vegetation was undertaken using the multivariate analysis DECORANA (Hill 1979), which is an ordination technique giving an index of similarity of the quadrats in terms of plant species composition. The dominant plant species, *Calluna vulgaris*, was omitted from some of the analyses to reduce the similarity of the vegetation quadrats and to emphasize changes in plant species composition. There was a consistent change in the plant species composition near the road, with an increase in the cover abundance of the grass *Molinia caerulea* and a decrease in the abundance of lichen species, particularly of the lichen *Cladonia impexa*. The regression of DECORANA score on distance from the road at each site was significant (Table 1).

Calluna plants growing near the road showed significantly greater annual growth increments and nitrogen concentration (2-way ANOVA $P<0.001$ in each case), the mean results being a shoot length of 61 mm, and 1375 mg/100 g nitrogen at 25 m from the road but only 41 mm and 1295 mg/100 g at 80 m from the road. There was no significant difference in tissue phosphorus concentration of *Calluna*. *Molinia* plants also tended to be larger near the road (2-way ANOVA P<0.05); the mean height being 406 mm 12 m from the road and 352 mm 80 m from the road. There was no significant difference in tissue nutrient concentration of *Molinia* with distance from the road. Clumps of the lichen *Cladonia impexa* were fewer and smaller near the road, but clump size was very variable and the trend was not significant. The lichen was shorter, and had fewer branches near the road (ANOVA P<0.01).

The carbon content of the soil decreased near the road, the mean value being 29% at 10m from the road, and 40% at 80 m from the road (2-way ANOVA P<0.001). The carbon:nitrogen ratio decreased near the road, from 28·0 at 80 m from the road to 24·7 at 10 m from the road (2-way ANOVA P<0.001), as did the carbon:phosphorus ratio (118 at 80 m from the road, 86 at 10 m from the road; 2-way ANOVA P<0.001).
From plots of change in the soil and vegetation with distance from the road, (e.g. Fig. 2), and with the aid of regression analyses, it was evident that the changes in the heathland habitat were detectable up to 200 m away from the road.

Table 1.
Summary of trends in plant species composition of heathland adjacent to a road, revealed by DECORANA.
The regression of DECORANA score on distance from the road is presented for either a linear or a multiplicative (mult) regression model.
Probability levels for the coefficient of determination (r^2) are indicated as NS (P>0.05), * (P<0.05), ** (P<0.01) or *** (P<0.001).
Species trends are indicated by + for species with DECORANA scores which indicate increased abundance near the road, and - for species with scores indicating a decline near the road.

		Decorana		Regression		Species Trends		
Site	Data Used	Axis	Eigenvalue	Model	r^2	*Molinia*	*Cladonia sp.*	*C. portentosa*
all	complete	1	0.272	linear	0.12***	+	-	-
	complete	2	0.195	mult.	0.10***	+	-	-
site 1	complete	3	0.112	linear	0.10 NS	+	-	-
	no 80m	2	0.134	linear	0.21 NS	+	-	
	no Calluna	2	0.250	linear	0.22*	+	-	-
	no moss	2	0.176	linear	0.41***		-	-
site 2	complete	1	0.285	mult.	0.19**	+	-	-
	no 150m	1	0.290	linear	0.55***	+	-	-
site 3	complete	1	0.211	mult.	0.17**	-	+	-
	no Ericaceae	3	0.116	linear	0.14*	+	-	-
site 4	complete	1	0.320	linear	0.12*	+	-	-
	no 200m	2	0.253	linear	0.36***	+		-
site 5	complete	1	0.284	linear	0.14**	+	+	-
	no Calluna	1	0.424	linear	0.16**	+		-
	no 200m	1	0.299	mult.	0.20**	+	-	-

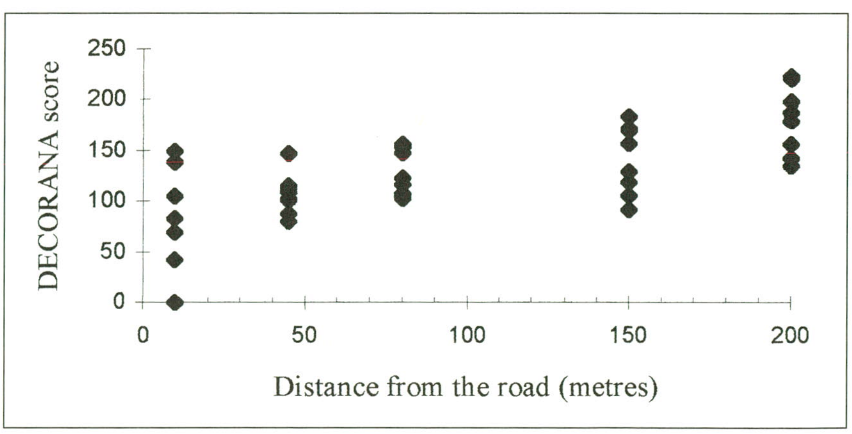


Figure 2.
Change in plant species composition of heathland with distance from the road. The y-axis is the detrended correspondance analysis (DECORANA) score of 50 cm x 50 cm plant cover quadrats. (Regression data: r^2=0.37; P<0.001)
The quadrat score is the weighted mean species score.
Low scoring species included the grasses *Molinia caerulea* and *Festuca rubra*. High scoring species included *Calluna vulgaris* and the moss and lichen species.

The impact of arable agriculture upon adjacent chalk grassland
There was a consistent change in the plant species composition of the grassland within 100 metres of a field. The grass species *Bromus erectus* and *Arrhenatherum elatius* increased in abundance near the fields along with species indicative of disturbed ground such as *Daucus carota*, and arable weeds such as *Taraxacum spp.* and *Plantago spp.* Forbs such as *Filipendula vulgaris*, *Helictotrichon pratense*, *Pimpinella saxifraga* and *Helianthemum nummularium*, which are valued members of the chalk grassland community, decreased in abundance near the fields (Fig. 3).

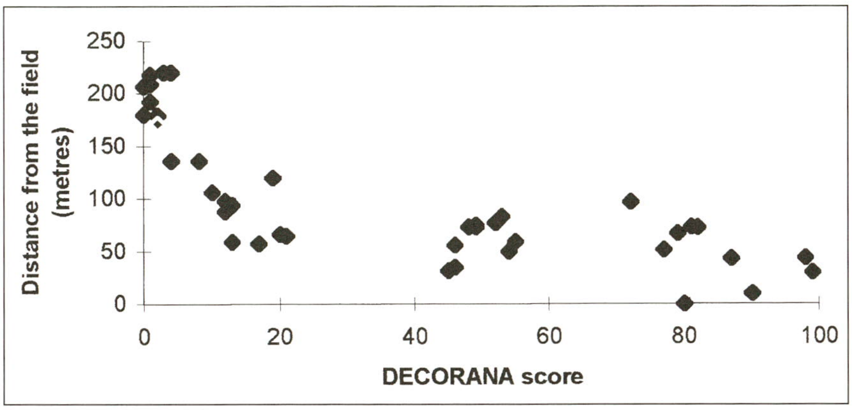
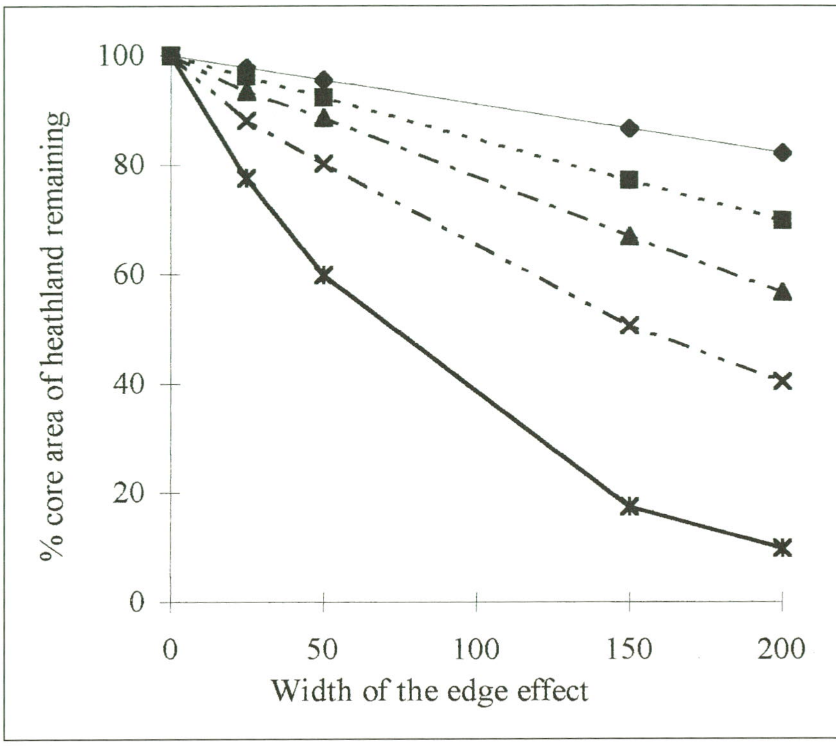


Figure 3.
Change in plant species composition of chalk grassland with distance from an arable field.
The y axis is the detrended correspondance analysis (DECORANA) score of 50 cm x 50 cm plant cover quadrats. (Regression data: r^2=0.57; P<0.001)
The quadrat score is the weighted mean species score.
High scoring species included *Bromus erectus, Arrhenatherum elatius* and *Daucus carota*.
Low scoring species included *Filipendula vulgaris, Pimpinella saxifraga* and *Helianthemum nummularium*.

Both *Bromus erectus* and *Centaurea nigra* grew significantly larger and more luxuriantly near the field, the mean biomass of plants beside the field being double that of plants 80 - 120 m from the field edge (2-way ANOVA P<0.001).
The mean carbon content of the soil decreased near the field, being 9% in the nearest sampling area, and 12% at 80 -120 m from the field edge (2-way ANOVA P<0.001). The carbon:nitrogen ratio ranged from 8 to 11, and showed no consistent trend with distance from the field. The mean carbon:phosphorous ratio decreased significantly near the field (2-way ANOVA P<0.001), being 65 in the nearest sampling area, and 80 at 80 - 120 m from the edge of the field.

Investigation into the implications of edge effects for nature conservation
The degree of loss of core area of habitat fragments increases with an increasing degree of habitat fragmentation (Fig. 4). The rate of loss of core area with increasing edge effect is constant providing that every fragment retains some core area. Whereas in 1759, all 7 extant heathland fragments contained large amounts of core area, in 1978 the degree of fragmentation was such that, assuming a 200 metre edge effect, only 11 out of the 435 extant heathland fragments retained any core area. The proportion of core area is shown to be drastically reduced by habitat fragmentation; the 7 large fragments in 1759 would have been 82% core area, but the heath was further fragmented to create 435 fragments, as little as 10% of that heath could be pristine core habitat.

Figure 4.
The percent core area of heathland remaining in the Poole Basin, U.K. at various dates, with increasing fragmentation and allowing for edge effects of up to 200m.
Key: ◆ = 1759, 7 fragments;
■ = 1811, 13 fragments;
▲ = 1934, 17 fragments;
× = 1960, 97 fragments;
✱ = 1978, 435 fragments.



Discussion

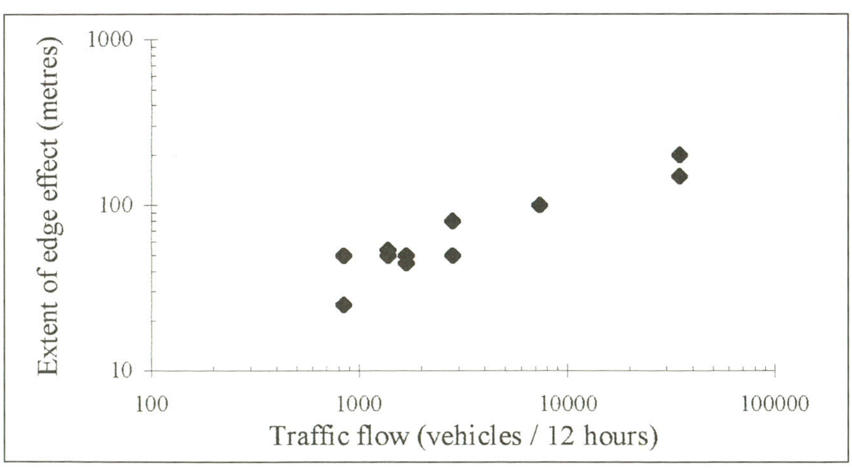
There were changes in plant species composition, plant performance and soil nutrient levels near to a road or an arable field. The competitive grasses *Molinia caerulea* and *Bromus erectus* increased in abundance and vigour, whereas the cover abundance of other plants such as *Calluna vulgaris*, *Helictotrichon pratense*, and *Helianthemum nummularium* decreased. The vascular plants had increased rates of growth near the edge of the habitat; in contrast the lichen *Cladonia impexa* grew less luxuriantly, and there was a decline in the diversity of the lichen flora near the road. These changes in the vegetation near the edge of the habitat are due to pollutants from adjacent land-use. The organic content of the soil decreased near the edge, which in view of the increased biomass produced by enhanced plant growth at the edge of the habitats, suggests an increased rate of decomposition and nutrient cycling. The soils near the arable field had a significantly raised carbon:phosphorus ratio, showing that the primary edge effect from the field was the phosphorus input. Further work has shown that these grasslands are phosphorus limited (Angold 1992), so the drift of phosphorus from agricultural fertilizers may be expected to have a profound effect upon competitive relations and the structure of the community, similar to that produced by the addition of nitrogen to the nitrogen-limited Dutch chalk grasslands (Bobbink 1991). In contrast, the primary edge effect generated by the road was a nitrogen input, demonstrated by the increased carbon:nitrogen ratio of heathland soils near a road. Oxides of nitrogen, which reached a flux of 60 ppb/hr near the road, were shown to exceed the winter critical level of 20 ppb/hr, which may cause damage to vegetation (Anonymous 1990); and many species of lichen are particularly vulnerable to atmospheric pollution (Ferry et al. 1973). Although all vascular plant species studied showed a fertilization effect, with improved performance near the edge of the habitat, more competitive plant species such as *Molinia* and *Bromus* are capable of greater increases in nutrient turnover and productivity than other species which maintain a relatively constant nutrient use efficiency (Aerts & Caluwe 1989, van der Hoek 1987).

This explains why the *Calluna* showed an increase in tissue nutrient concentration whereas the *Molinia*, which responds by greater productivity, maintained a constant tissue nutrient concentration. The soils nearer the road had greater carbon:nitrogen and carbon:phosphorus ratios, and a decreased organic content, which indicate an increase in the availability of nutrients near the road. When soil nutrient levels are high, *Molinia* litter is enriched in nutrients and decomposes more rapidly than *Calluna* litter (Aerts & Caluwe 1989), and *Molinia* tends to become dominant (Aerts 1989). Atmospheric deposition of nitrogen in the Netherlands has been implicated in the decline of the ericaceous shrubs and the increasing dominance of grasses in Dutch heathlands (Bobbink et al. 1990).

The results of this study show that a positive feedback is established near roads, in which nitrogen from vehicle exhausts causes increases in the growth rate of *Molinia*, resulting in a higher rate of decomposition (reflected in the lower carbon content of the soil), and greater nitrogen availability (reflected in the decrease in carbon:nitrogen ratio) in the soil, which further increases the rate of growth of *Molinia*. This results in an edge effect, a gradient of change in the heathland vegetation extending from the road at least 200 metres into the heathland habitat to either side of the road.

A study (Angold 1992) of the heathland adjacent to other, more minor roads in the New Forest has demonstrated that the extent of the edge effect is closely correlated with the amount of traffic the road carries (Fig. 5), and therefore with the amount of atmospheric pollution from vehicle exhausts. (There was no measurable effect of prevailing wind direction upon extent of the edge effect at these study sites.) A similar positive feedback of phosphorus addition has been found in chalk grassland adjacent to arable fields, with an edge effect extending up to 100 m from the field edge.

Figure 5.
The relationship between traffic volume and the extent of the edge effect in heathland in the New Forest, Hampshire, UK .
(r^2=0.71; P<0.001).



The scale of edge effect shown by this study is similar to other work. Heavy metals are shown to accumulate in lichens during transplant experiments up to 70 metres from a road (Tuba & Csintalan 1993, Semadi & Deruelle 1993), and the disturbance to large mammals from road traffic can extend hundreds of metres from the road (It should be noted that this study has focused on the plant community, and that edge effects on animals sensitive to noise pollution may be far greater than those shown here: McLellan & Shackleton 1988, Janzen 1986).

The case study of heathland fragmentation in the Poole Basin was used to demonstrate the implications of the edge effect for nature conservation. In 1759, when there were seven large heathland fragments (33603 hectares of heathland in total), the percent loss of area would be only 18% for an edge effect of 200 metres. In 1978, when there were 435 heathland fragments with a total area of 5731 hectares, 90% of the area regarded as heathland would be disturbed edge habitat, and of the remaining 10%, the core area in some fragments may be too small to support the full heathland community (see Webb et al. 1984; Webb & Vermaat 1990). The core area of undisturbed habitat in a fragmented landscape may thus be substantially smaller than it initially appears. This study demonstrates the importance of a buffer zone to absorb edge effects from land use adjacent to protected habitats. It should be noted that edge habitat is often of great value for conservation, e.g. Dowdeswell (1987) discusses the importance of road verges for conservation; but it is not 'core area' and should not be included in calculations of the size of a given habitat fragment for nature conservation (Laurance & Yensen 1991). The place for edge habitats is not in the reserve, but in the buffer zone, where its importance for edge and generalist species will be undiminished, and where it will protect the core area from edge effects (Harris 1984).
The use of buffer zones to protect areas of conservation importance should be incorporated into environmental impact assessments as part of the mitigation measures to reduce the impact of development on adjacent habitats. The width and type of buffer zone necessary may vary according to both the habitat and the development under consideration. The width of an existing buffer zone would need to be reconsidered following applications for any subsequent new developments in the region of a protected area, and an extension of the buffer zone may be necessary in some cases.
The benefits of conservation buffer zones need to be considered routinely in planning procedures, the important aspects being:

- the area of the affected habitat, and its conservation importance
- the predicted long term edge effect from the proposed development
- the area of land required to absorb this edge effect and thus prevent degradation of the habitat.
- further research is needed into the efficiency of measures such as artificial barriers or embankments reducing the area of land required for a buffer zone.

Acknowledgements

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Impact of landscape changes on the ground beetle fauna (Carabidae) of an agricultural countryside

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Abstract

A landscape section (100 km²) of an intensely cultivated countryside in Germany was selected to analyse effects of landscape changes (quantified by GIS) on one group of invertebrates, the Carabidae. The carabid fauna was studied between 1990 and 1994 by setting pitfall traps at altogether 36 sites.
Three main periods of landscape change are distinguished for the last 150-200 years. High loss and fragmentation of old forests but also a dramatic decrease in the number of ponds are among the major trends detected. Species of habitats which are isolated or unstable under natural conditions are affected by landscape change only to a minor extent.
The strongest fragmentation effects were registered for species of ancient woodland. The occurrence of the threatened cavernicolous species Laemostenus terricola appears to depend on both the presence of rabbit burrows and the connectedness of old sandy embankments. The consequences for nature conservation are discussed.

Introduction

Infrastructural changes are continually causing loss and fragmentation of natural and semi-natural habitats in agricultural regions of Europe (Agger & Brandt 1988, Burel & Baudry 1990, Ihse 1995). Measures of land consolidation and road construction still have great impacts in such areas.
Effects of habitat isolation and fragmentation or of different spatial arrangements and proportions of hedges, woodlots, forest- or heath fragments on species distribution, dispersal and survival are major subjects of many current ecologial studies (van Dorp & Opdam 1987, Burel 1989, de Vries & den Boer 1990, Opdam 1990, Andren 1994, Gruttke & Kornacker 1995, Vermeulen 1995).
Good knowledge of the consequences of landscape change is a necessary condition for the development of measures to improve landscape structure in order to maintain a rich and typical fauna and flora in agricultural regions.
The main objectives of the present study are:

- to quantify reductions of those habitat types in a cultivated landscape which have been most affected by landscape changes during the past 100-150 years,
- to analyse and describe the impact of landscape change on the ground beetle fauna *Carabidae*, as representative of other groups of soil surface dwelling arthropods,

- to find out species and/or ecological groups of species which are most threatened by landscape changes today, and
- to draw conclusions from the results for setting up priorities for landscape planning and nature conservation.

Study area, materials and methods

The investigation was carried out in an intensely cultivated agricultural landscape about 30 km West of Bonn, Germany. The main crops in this area are sugar beet, barley and wheat. According to Trautmann (1991) the natural vegetation had been beech- or oak-hornbeam forest with a significant proportion of lime trees *Tilia cordata*. Most of todays forest remnants belong to last mentioned vegetation type, named Stellario-Carpinetum.
The climate is mild and relatively dry in summer with an average precipitation of 570 mm/year and a mean temperature of 9.6 °C.
Within a landscape section of about 100 km^2 the amount of woodland, grassland, agricultural fields, human settlements and of linear features like roads, embankments, ditches, hedges and streams was quantified on the basis of present and historical maps (of 1845, 1893, 1938 and 1989) using the GIS-system Arc-Info. Additionally, vegetation cover and soil parameters were registered in the field.
The carabid fauna was studied between 1990 and 1994 by setting pitfall traps - usually 5 (in 1994 8) as a standard set (upper diameter: 4.5 cm; preservative: ethyleneglycol) - at altogether 36 sites: 26 wooded habitats of different age and size (7 young hedges and woods: Aa, Ab (motorway embankments), Fu, U2, U5, U6, U9; 10 medium aged to old woodlots and hedges Ah, W, Ra - Re, Nb, Gi, Ki; 9 old forest remnants Mh, Ba, Bb, Wa, Wf); 6 meadows (Suk1 and 2: dry habitat strips with a wet ditch running through the middle; mA1 - mA4: young meadows, medium dry to moist), one small wet area (Wo) partly with open soil partly with shrubs and a reed and 3 agricultural fields (F1 - F3). To complete the species list of the study area results from hand samplings and captures in other parts of arable fields were considered, as well.
Nomenclature of Carabidae follows Freude et al. (1976) and Lohse & Lucht (1989). For the ecological classification of species, mainly the publications of Barndt et al. (1991) and Turin et al. (1991) were used. Statistical calculations, ordinations and clusterings were done with the progamm NCSS 5 (Number Cruncher Statistical System, by Dr. Jerry L. Hintze, Kaysville, Utah USA).

Results and discussion

Analysis of landscape change
For a long time (at least 200 years), the dominating landscape element of the studied region has been farmland. More than 80% of the total area is covered by arable fields (in 1893: 86%, in 1989: 82%). Only about 10% of the area remains for all other types of habitats.
For the last 150-200 years three main periods of landscape change can be distinguished. The first period (1845 until about 1920) is characterised by a high loss and fragmentation of old forest habitats (compare Fig. 1).

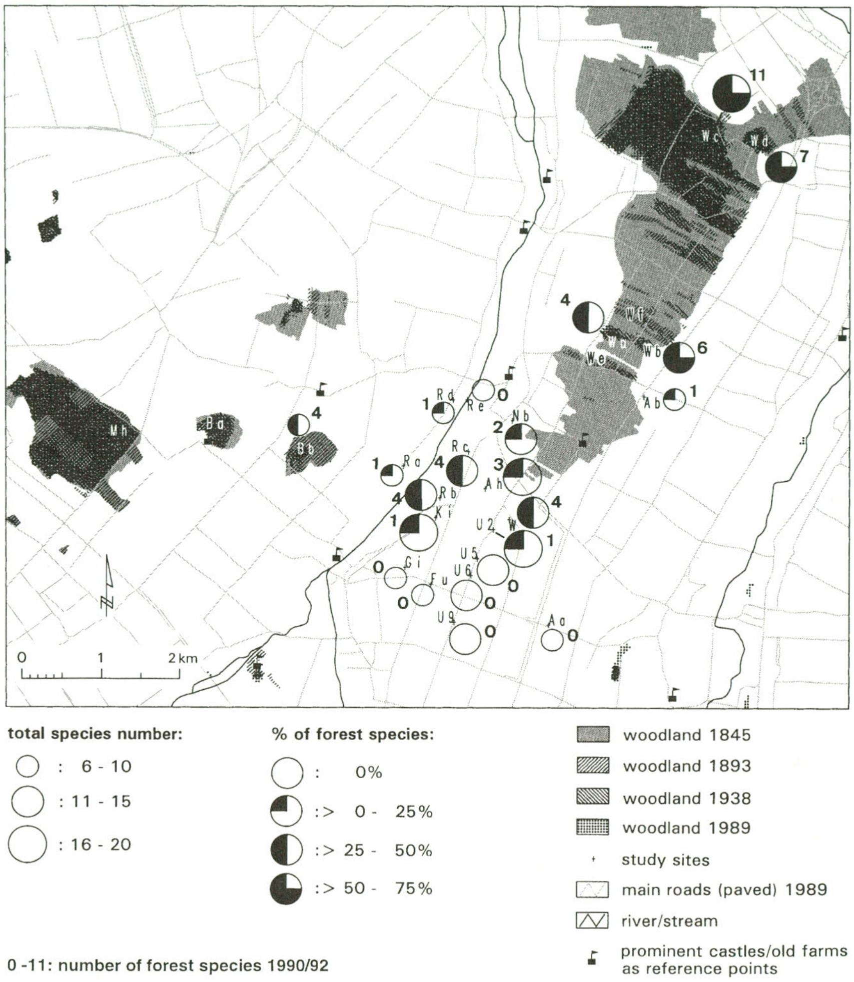


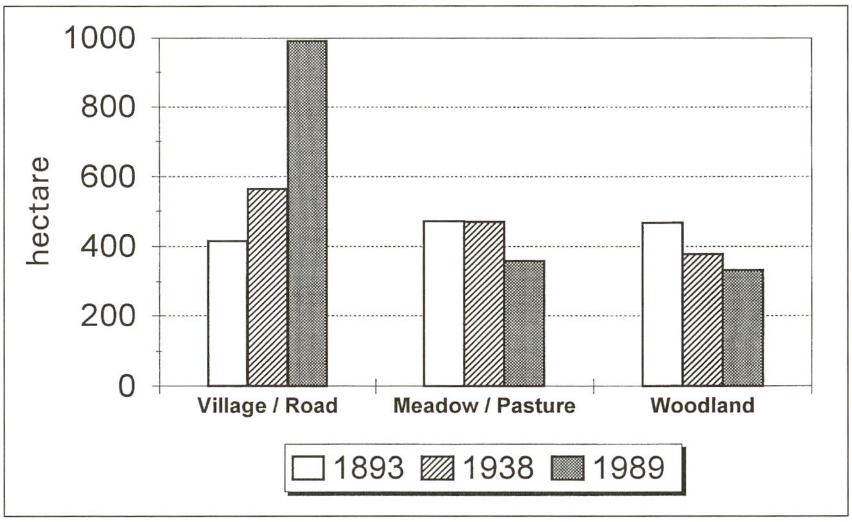
Figure 1.
Total number of carabid species, percentage and number of forest carabid species at wooded study sites in 1990/1992 and decline of woodland area between 1845 and 1989.
The darkest hatched areas represent (with one exception) the forest distribution today, because nearly all forest areas had only been reduced in size during that period.

Between 1845 and 1893 the area of woodland decreased by more than 56%. The fragmentation process slowed down afterwards. Today's forest area represents 31% of its initial size in 1845, which was about the same as in 1800.

The second period starts with the first land consolidation in the late twenties. During this time the structure of the rural road network and the size of fields was completely changed. The length of unpaved rural lanes increased from 340 km in 1893 to 507 km in 1938, which may have improved living conditions for species of meadow habitats because of a better connectedness of their habitats by grassy road margins. Negative isolation effects caused by unpaved lanes are not very probable in such a type of open landscape (cf. Schell & Mader 1989). The amount of grassland remained stable during this period, but the number of field ponds was reduced for more than 50% (in 1893: 109, in 1938: 53).
The second land consolidation in the seventies - starting the third period - introduced modern infrastructure into the agricultural landscape, with large

fields, altogether less road length but more of it paved (in 1938: 81 km, in 1989: 263 km) and a new motorway (16 km) crossing the area. Small habitats like field margins, ditches or ponds were dramatically reduced.
In addition to the loss of woodland mentioned before, the area of grassland decreased for about 25% during this time. Fallow land on former gravel or sand pits was one of the few habitat types expanding in this period.
Figure 2 summarizes major trends for change between 1893 and 1989 for three land use types.

Figure 2.
Changes in area within one century of three main types of land use.



Development of carabid fauna
An overall number of 101 carabid species was recorded in the study area (Table 1, listing diagnostically significant species). Hierarchical clustering of the carabid samples from 1990 -1992 revealed eight major groups of sites with distinctly different faunistic composition (Fig. 3). Ordinations resulted in similar but less clear differentiated clusters.

Figure 3.
Hierarchical clustering (type: complete linkage) of study sites according to the abundances of carabid species (per five traps) of samples from 1990, 1991 and 1992 (four sites studied only in 1994 are not included).

Cluster groups:
- I strips of dry meadows and young shrubs;
- II old forest remnants;
- IIIa-c wooded banks, woodlots and hedges of different age,
- IV arable fields;
- V single wet habitat surrounding a pond;
- VI meadows medium dry to moist.

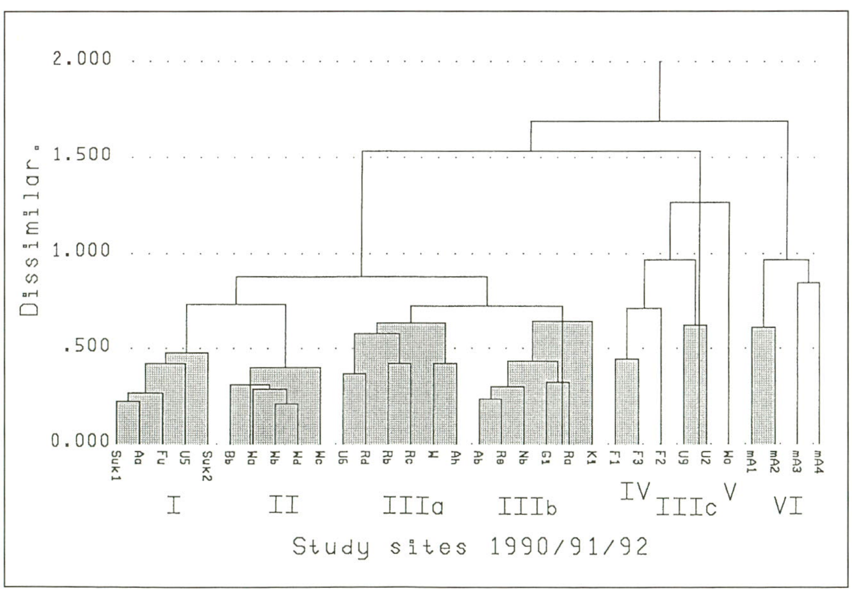


Table 1.
Distribution and frequencies of diagnostically significant species ordered according to their spatial occurrence in 1990, 1991 and 1992 (160 sampling days).

Abundance classes (individuals/20 traps):
1: >0 to ≤2
2: >2 to ≤4
3: >4 to ≤8
4: >8 to ≤16
5: >16 to ≤32
6: >32 to ≤64
7: >64 to ≤128
8: >128 to ≤256
9: >256 to ≤512
10: >512.

Habitat classification:
M: matrix habitat;
F: frame habitat;
FW: framework habitats;
IS: island habitats.
(further explanation in the text).

Species	Abundance classes: Arable fields	Strips of dry meadows and shrubs	Meadows medium dry to moist	Wet area around pond small reed	Wooded banks woodlots hedges	Old forest remnants
Calathus cinctus	8	1	2		1	1
Poecilus cupreus	1		5		1	1
Calathus fuscipes	4	1	6		3	
Carabus monilis	2	3	1		2	
Bembidion lampros	4	1	9	4	1	
Pterostichus strenuus	1	1	2	6	1	
Clivina fossor	3	1	4	6		
Amara bifrons	2	2	3	3		
Agonum mülleri	2		4	4		
Harpalus affinis	2		7		1	
Harpalus rufipes	1		9		1	
Amara aenea	1		5		1	
Bembidion obtusum	1				1	
Zabrus tenebrioides	4		2			
Calathus ambiguus	4		2			
Demetrias atricapillus	3					
Trichotichnus nitens	1					
Amara lunicollis		3	4	5	1	2
Harpalus rubripes		6	2		2	
Amara convexior		3	3		1	
Carabus auratus		2	2		2	
Syntomus truncatellus		1			1	
Harpalus puncticeps		4				
Agonum moestum		1		9		
Anisodactylus binotatus			7	5	1	
Notiophilus palustris			5	3	2	
Dyschirius globosus			3	6	1	
Laemostenus terricola			4		9	2
Nebria brevicollis			2		2	2
Harpalus latus			1			2
Harpalus tardus			6		3	
Amara aulica			5		1	
Acupalpus meridianus			4			
Amara apricaria			3			
Stenolophus teutonus			2			
Bradycellus verbasci			2			
Poecilus versicolor			2			
Harpalus distinguendus			1			
Pterostichus vernalis			1	6		
Platynus obscurus				10		
Pterostichus nigrita				9		
Agonum fuliginosum				9		
Bembidion dentellum				8		
Platynus albipes				7		
Pterostichus rhaeticus				7		
Agonum pelidnum				6		
Pterostichus minor				5		
Bembidion biguttatum				5		
Stenolophus mixtus				5		
Agonum viduum				5		
Elaphrus cupreus				4		
Bembidion assimilis				4		
Acupalpus dubius				3		
Calathus rotundicollis				4	6	5
Notiophilus biguttatus					3	1
Amara communis					2	1
Pter. oblongopunctatus					2	5
Leistus rufomarginatus					1	1
Harpalus nitidulus					2	
Dromius linearis					1	
Dromius melanocephalus					1	
Notiophilus germinyi					1	
Carabus problematicus						7
Abax parallelepipedus						6
Platynus assimilis						3
Harpalus quadrimaculatus						2
Pterostichus madidus						2
Abax parallelus						1
Amara ovata						1
Notiophilus rufipes						1
Cychrus caraboides						1
Mean species no. per site	23	12	32	38	13	12
Standard deviation	2,5	4,4	6,5	-	3,9	2,7
Habitat classification	M	FW	F	IS	FW	IS

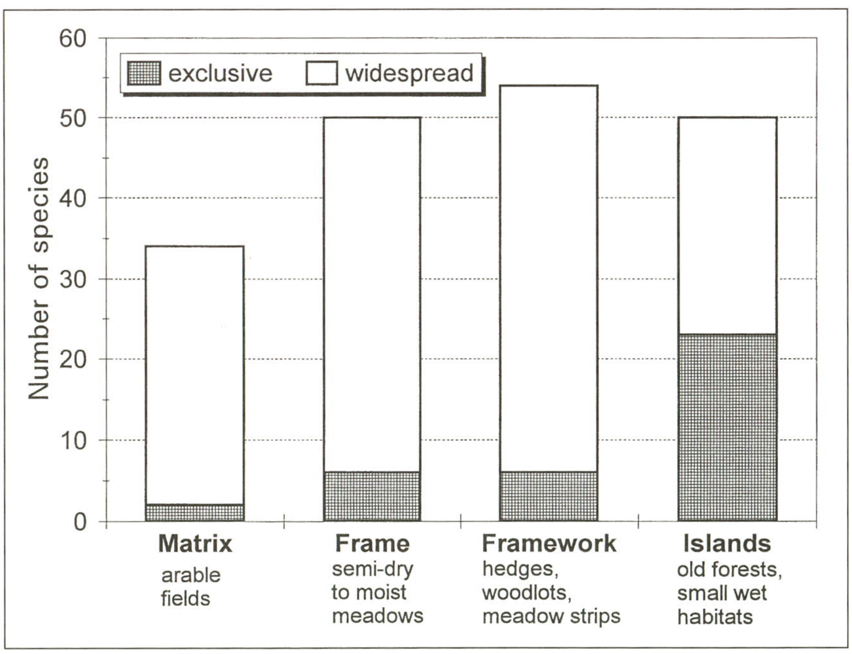
Widespread species (19) not listed above: Amara familiaris, Amara similata, Asaphidion flavipes, Badister bullatus, Bembidion quadrimaculatum, Bembidion tetracolum, Calathus melanocephalus, Carabus nemoralis, Leistus ferrugineus, Loricera pilicornis, Panagaeus bipustulatus, Platynus dorsalis, Pterostichus melanarius, Pterostichus niger, Stomis pumicatus, Synuchus vivalis, Trechoblemus micros, Trechus obtusus, Trechus quadristriatus.
Another 11 species found only in handsamples or caught at other sites of the study area or in other years are not listed here.

According to their spatial and structural characteristics and their functional role in the landscape, these eigth habitat groups were assigned to four classes of landscape elements:

1. the matrix habitat (M): arable land (cluster IV in Fig. 3);
2. frame habitats (F): grasslands, mostly in river valleys (the sites studied were young meadows, cluster VI);
3. framework habitats (FW): the network of field margins and meadow strips with single young and old woodlots and hedges (cluster I and III a-c in Fig. 3);
4. island habitats of specific type (IS): wet areas around ponds with small reeds (cluster V) and old forest fragments of different size (cluster II).

Figure 4 clearly shows that the number of species living in the matrix habitat is much lower than respective numbers in all other classes of habitats.
A total of 37 species was only found in one of the habitat types distinguished. The highest number (23) of these so called "exclusive" species occurred in habitat islands (14 hygrophilic species in one wet patch (Wo), 9 species in old forests). To enable long-term survival of last mentioned group of "exclusive" species in the agricultural landscape population exchange between the habitat patches has to be guaranteed (cf. den Boer 1990, Opdam 1990).

Figure 4.
Total number of species and number of "exclusive" species caught in at least three specimen (except in the single wet habitat where a limit of two specimen was set) per class of landscape element (Matrix, Frame, Framework, Island). Further explanations in the text.



Impact of fragmentation on the carabid fauna
Fragmentation and landscape change are mostly a problem for species which do not belong to the matrix habitat type farmland. Species living in arable fields are usually widespread and eurytopic (see Table 1) and habitat size or isolation are not limiting factors for their survival. Nevertheless modern agricultural practices - mainly fertilization and pesticide use - have affected the occurrence of some species typical for agricultural fields in former times (Heydemann & Meyer 1983). One of these species - *Carabus auratus* - is

nowadays completely restricted to marginal, more or less open habitats of the study area (cf. Gruttke 1992).

Carabid species of habitats which are isolated or unstable under natural conditions, for instance species of young successional stages or of banks of rivers and ponds, are affected to a minor extent, as well. Numbers of typical species at the remaining suitable sites (e.g. a total of 19 hygrophilic species at the wet study site Wo) are still as high as in comparable landscapes nearby with higher proportions and numbers of wet habitats, for instance (Koch 1977, Brechtel & Riedel 1989).
A special case is the cavernicolous beetle *Laemostenus terricola* which is mainly living in rabbit burrows. As shown in Figure 5 the distribution of this species at landscape scale is to a great extent dependent on the occurrence of sandy soils. In contrast to results at habitat scale (Gruttke 1994), no significant correlations between numbers of rabbit burrows and beetle catch numbers were obtained at landscape scale. However, beetle abundances were highly correlated to the length of historically connected sandy embankments (Spearman rank-correlation rs = 0.9350; α: < 0.001), even when they are devided by a road, today.

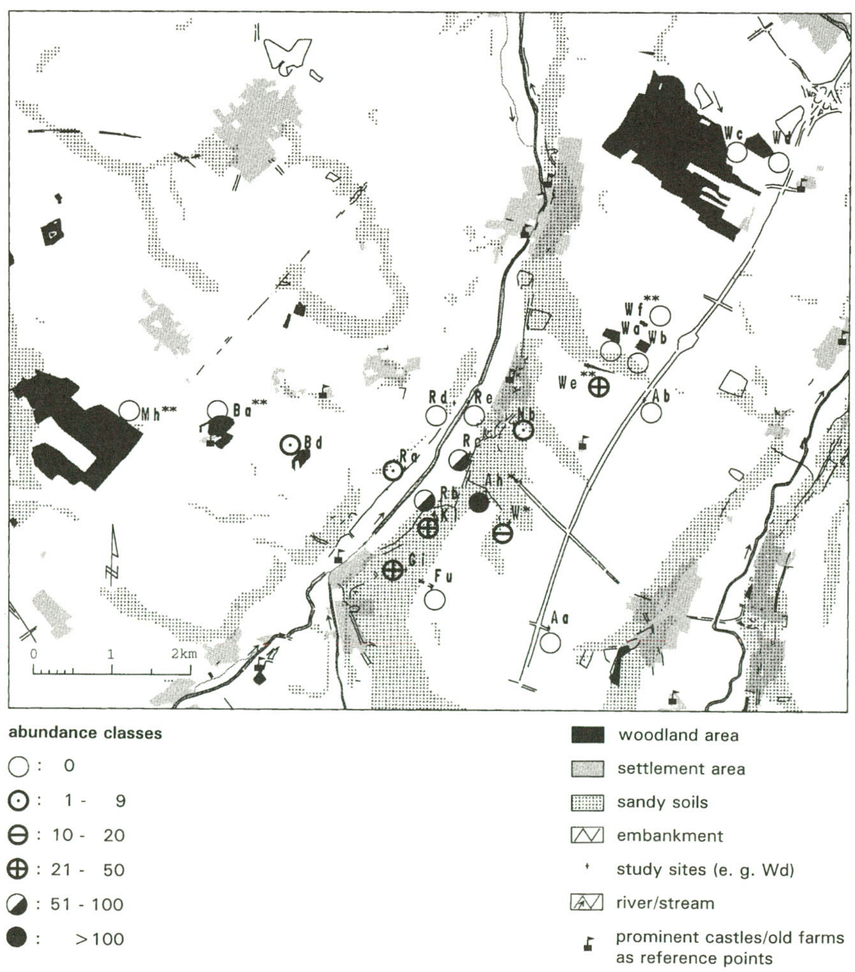


Figure 5.
Distribution and abundance of the species *Laemostenus terricola* at sites with rabbit burrows in relation to the occurrence of sandy soils and embankments.

Most of the sites were sampled in 1992;
*: results of woodlot W and old hedge Ah from 1990;
**: results of additional forest sites Mh, Ba, We and Wf from 1994).

The strongest fragmentation effects were observed for species of stable habitats, in the study area only old woodlands. Some of these old forest patches became "islands" more than 200 years ago (Fig. 1: Ba and Bb, in the western half), others had been split off larger forests only 50 to 150 years ago (Fig. 1 Wa, Wb, Wd, We and Wf, in the eastern part). Broadening of roads and the construction of a motorway which separates all forests studied from a large woodland area in eastern direction, increased isolation of the remaining old forest fragments during the last decades.
The results on the carabid fauna show that isolation of forest habitats today (measured as distance to next forest > 1 ha or > 100 ha) is the most important parameter determining composition and number of forest carabid species (Tab. 2, Fig. 6). To some minor extent and particularly for certain species, the actual size of forests also seems to have some influence.
An example is the stenotopic beetle *Abax parallelus* which is only occurring in old forest remnants of a size of at least two hectare. Other factors examined (e.g. soil pH, cover of vegetation within forests) did not correlate significantly with the number of forest carabid species.

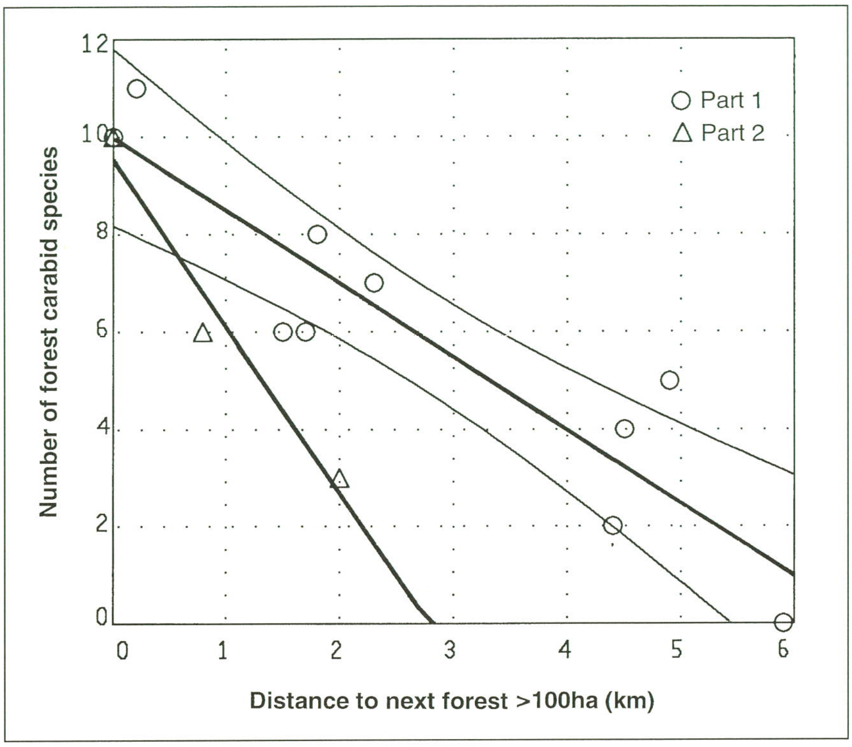
Table 2.
Rank-correlations (rs) and partiell rankcorrelations (prs) between number of forest carabid species in 1994 (15 in total) and the three main landscape parameters.
(Significance levels (n= 13):
$\alpha = 0.001$: rs ≥ 0.7917
$\alpha = 0.005$: rs ≥ 0.6978
$\alpha = 0.01$: rs ≥ 0.6429
$\alpha = 0.05$: rs ≥ 0.4780).

parameter	no. forest species	no. forest species	no. forest species	no. forest species
correlation / parameter	rs	prs controlled for area woods	prs controlled for distance next wood > 1 ha	prs controlled for distance wood > 100 ha
area woods ($\log_{10}$ of area)	0,7577	----------	0,3739	0,2027
distance next wood > 1 ha	-0,8496	-0,6619	-----------	0,4668
distance next wood > 100 ha	-0,8117	-0,4815	-0,2024	----------

Some of the effects found can also be attributed to fragmentation history. The connectedness of forest patches and thus connectivity of populations in the past appear to be of significance for the present distribution of forest carabids, as well. The decrease in number of forest species with increasing distance to the next large forest shown in Figure 6, is stronger for the western forest complex in which the islands Ba and Bb had been isolated for a very long time than it is in the eastern forest complex which was fragmented more recently.
About half of the forest carabid species recorded (8 of 15) were restricted to old forest remnants and did not move into young habitats (cf. Assmann 1994). However, metapopulation processes seem to stabilize populations of a number of other forest species, some of which also colonized interconnected medium aged woods. One example for this is *Pterostichus oblongopunctatus*, a species which was caught outside old forest fragments only in a complex of more or less connected woodlots and hedges (sites W, Ah, Rc and Ni, see Fig. 1).

Figure 6.
Decrease of numbers of forest carabid species (in 1994) with increasing distance to next old forest larger than 100 hectare: a comparison between eastern (W, Ah, Gi, Rb, Wa, Wb, Wc, Wd, We, Wf) and western (Ba, Bb, Mh) complexes of medium aged to old woods (compare Fig. 1).

Regression lines (bold lines) and confidence limits of mean for eastern part (α= 0.05);
part 1: eastern complex, regression: y= 9.98 - 1.5x;
part 2: western complex, regression: y= 9.53 - 3.4x.



Roads are very strong barriers for a number of forest carabid species and thus impede dispersal activities to a high extent (Mader 1979). In the present study, nearly all forest species were missing in small, completely isolated wooded habitats (Fig. 1). Even old but very isolated and small woodlots as site Gi are not colonized by forest species. Not even one specimen had been found at this site during two years of investigation .

Conclusions

With regard to landscape management and nature conservation the following conclusions are drawn from the results:

1. High species richness is found in some hedges and small woodlots, young grassland and meadowstrips, all of them "framework" habitats of the agricultural landscape. But only a minority of these species can actually be regarded as endangered, because most of them are eurytopic and thus able to live in or at least pass through different types of habitats (often including arable fields).
2. Old large interconnected embankments with hedges and woodlots on sandy soils are the preferred habitat of *Laemostenus terricola*, a species regarded as endangered in several parts of Europe (Desender & Turin 1989). For maintaining this species in the cultivated landscape such habitats have to be preserved, although or because they are also the preferred habitats of rabbits.
3. The highest number of individuals and of species in total, but also of stenotopic species, has been registered in a small wet habitat surrounding

a pond. For keeping biodiversity high in agricultural landscapes the conservation of such habitats should have priority. Nevertheless the carabid fauna of these wet habitats is not much endangered today. Most of the species living there are able to colonize adequate, newly installed wet habitat within a short time period because of their high dispersal power.

4. Among all habitat types studied, ancient woodlands and their carabid species were most affected by habitat fragmentation. Thus enlargement and connection of ancient forest remnants should have priority upon the plantation of new forests without any contact to old woodland. Patches of ancient woodland should not be subdivided by roads. The habitat quality of old forests for stenotopic species is markedly reduced below a size of about 10 ha (cf. Gruttke in press), e.g. by edge effects .
5. Even relatively small medium aged woodlots (minimum size 0.2 - 0.4 ha) can be effective as stepping stones for some - but not all - forest carabid species, as long as they are well interconnected (max. distance about 50-100m) (cf. Glück & Kreisel 1986). Old wide road banks (like study site Rb) can serve as corridors, too.
6. Young and isolated woodlots but also young embankments of roads are no adequate stepping stones for most carabid species affected by fragmentation. A minimum habitat age of about 50 years seems to be necessary.

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Assessing fragmentation of bird and mammal habitats due to roads and traffic in transport regions

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Keywords: habitat loss, disturbance, barrier action, casualties, modelling

Abstract

To tackle the problems caused by roads and traffic the Dutch government has introduced a new policy instrument for use at the regional level: the Transport Region. In this context, a computer model is being developed for evaluating policy measures. The environment module of this model must also take due account of the fragmentation of nature. This article describes a method for achieving this in the case of birds and mammals. Fragmentation by roads and traffic comprises the aspects of habitat loss, disturbance, barrier action and collisions. By linking

- *information on the size of roads and intensity of traffic,*
- *the occurrence of habitats and*
- *intervention-effect relationships with species as effect parameters,*

a quantitative indication can be given of the extent of habitat loss, disturbance and collisions. Because of the inaccuracy of the data on habitat location and owing to limitations of the computer model used, the extent of barrier action cannot yet be determined.

Introduction

The construction and (sharp rise in) the use of road infrastructure are giving rise to increasingly severe problems in terms of accessibility, mobility and quality of life in the Netherlands. These are leading not only to a deterioration in the quality of the human environment; nature is also suffering the consequences, particularly as a result of fragmentation (cf. Cuperus & Canters 1997).

Here, we define fragmentation as the loss and dissection of habitats and ecosystems as a result of human activities. We distinguish the following aspects of fragmentation due to roads and traffic:
1. loss of habitat by the physical presence of a road,
2. disturbance, e.g. by traffic noise,

3. barrier action, i.e. the separation of habitats, and
4. fauna casualties.

As an illustration of the Dutch government's growing concern about fragmentation as an environmental item, it is instructive to report several targets set by the Roads and Hydraulic Engineering Division of the Ministry of Transport and Public Works (Anonymous 1993):

- in 2000: 40% of the conflict points (reference year 1990) between the National Ecological Network (i.e. a continuous national ecological network connecting natural areas) and the Trunk Road Network (i.e. those roadways designated as 'main roads' in the Netherlands) are to be reduced;
- in 2010: 90% of the conflict points (reference year 1990) between the National Ecological Network and the Trunk Road Network are to be reduced.

Although the conflict points are still to be defined in greater detail, these objectives make it clear that, in addition to a need for further information about existing and future fragmentation, there is also a need for more information on suitable mitigating measures. These measures can best be generated and selected if a method is available that gives an indication of the fragmentation due to a road or complex of roads that is as complete and up-to-date as possible. Such a method can then be used to propose measures, estimate costs and set priorities.

Our purpose was thus to develop a method with which to chart the problems resulting from the fragmentation of nature en which could be implemented in a transport and traffic computer model for transport regions (see below). In this study we have concentrated on the impacts of fragmentation by roads and traffic on breeding birds and mammals. The reasons for focussing on these two groups were:
1. the objective to develop in the first instance a relatively simple method (more groups should have made the study too complicated),
2. the relatively wide body of knowledge about the effects of roads and traffic on birds and mammals, and
3. the existing experience in our institute with these both groups.

In this paper we report the most important findings; other results and further details of the study can be found in Cuperus & Canters (1997).

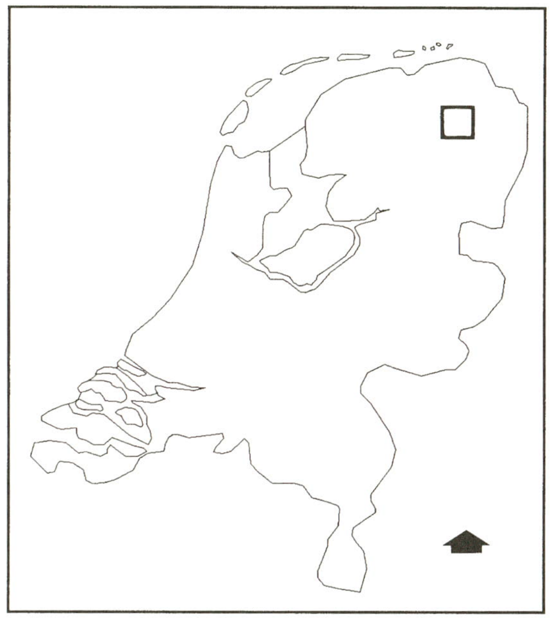
Methods and materials

Among the tools introduced by the Dutch government to achieve its transport policy objectives with regard to mobility, accessibility and quality of life is the so-called Transport Region (Anonymous 1990). A Transport Region is the area in and around one or more larger towns or cities that fulfil a regional pivot function. Such a region may vary in size from a province to a metropolitan district and generally encompasses the areas administered by several municipalities that are functionally related to one another.

The Groningen region forms a pilot area for doing research into the possibilities of the transport region. This region is situated in the north of the

Netherlands (Fig. 1) and comprises the towns of Groningen and Assen and their outskirts and the nine rural municipalities situated between them and has an aggregate area of approx. 700 km². The countryside consists mainly of open arable farmland in the north-east, grassland in the west and north-west - with in some places a relative abundance of tree-lined fields - and a semi-open landscape in the south with several fair-sized stretches of woodland.

Figure 1.
Location of the Groningen transport region in the Netherlands.



As part of this policy effort a computer model has been developed: the Transport and Traffic Evaluation Instrument, known by its Dutch acronym 'EVV'. The EVV model is designed for use in evaluating regional-level policy measures that are already in place and comparing alternative policy measures that may possibly be taken. This model contains detailed, stretch-by-stretch information on the roads in a given transport region and the traffic using these stretches of road. For our purpose, the most important information is: the location and width of each stretch, and the traffic intensity and speed limit in force on each stretch (information about the real speed is in the EVV model not available).

The information available on the actual occurrence of birds and mammals in a given transport region is often insufficiently detailed and comprehensive in terms of coverage. On the other hand, there is information available on the occurrence of ecotopes (incl. their area, but not their exact location!), viz. in the LKN database set up by the Netherlands Landscape-Ecological Mapping project (Bolsius et al. 1994, cf. Canters et al. 1991). This database is based on a continuous grid of 1x1 km cells, and each cell contains information on its abiotic and biotic components, including the aforementioned ecotopes. To the extent that they are relevant for all the transport regions in the Netherlands, we clustered the ecotope categories used in the LKN database into seven habitat categories (≈ landscape types) which, in the context of this project, are sufficiently distinctive to be used as a basis for the occurrence of birds and mammals: Arable farmland, Grassland, Heathland, Open Water, Marshland, Semi-open landscape and Woodland (i.e. Dunes and Mudflats as the still resting categories were left out, because of their relative small areas under the influence of roads and traffic in the Netherlands, especially in the transport regions). In calculating the area of a given habitat per kilometre grid cell we have corrected the open habitats Arable Farmland and Grassland for the presence of built-up area, woodland and 'linear green elements' (hedgerows, wooded banks, etc.), since in practice these ecotopes restrict the area of suitable habitat available for the species of these habitats.

For each aspect of habitat fragmentation and each habitat category we then selected the most sensitive species -from literature- for the aspects disturbance and collisions (for a review see: Cuperus & Canters 1997), subject to the criterion that the species should not be uncommon or rare (Table 1). It was assumed that the selected species did indeed occur in the habitat category in question. These species are actual present in the Groningen transport region (cf. SOVON 1987). In order to predict effects, we used the information found in literature on intervention-effect relationships.

Table 1.
The most sensitive, not uncommon species for the aspects 'disturbance' and 'collisions' for each habitat category.

aspects ▸ / ▾ habitat	DISTURBANCE	COLLISIONS
ARABLE FARMLAND	Skylark *Alauda arvensis*	Hare *Lepus europaeus*
GRASSLAND	Black-tailed Godwit *Limosa limosa*	Common Kestrel *Falco tinnunculus*
HEATHLAND	Skylark *Alauda arvensis*	(no species found)
OPEN WATER	(no species found)	Moorhen *Gallinula chloropus*
MARSHLAND	Common Cuckoo *Cuculus canorus*	Moorhen *Gallinula chloropus*
SEMI-OPEN LANDSCAPE	Common Cuckoo *Cuculus canorus*	Barn Owl *Tyto alba*
WOODLAND	Common Cuckoo *Cuculus canorus*	Hedgehog *Erinaceus europaeus*

On the basis of the information on the road stretches provided by the EVV database we were able to quantify the interventions leading to three aspects of habitat fragmentation: habitat loss, disturbance and collisions.
Because the EVV road stretches and the LKN kilometre grid cells use the same system of spatial coordinates, it was moreover possible to link the presence of the six habitat categories to the road stretches. Since the exact locations of habitats were unknown, in establishing such a linkage we assumed a uniform spatial distribution of relevant ecotopes per kilometre grid cell.

For example, 60 ha woodland in a kilometre grid cell was interpreted as 60% of the influenced area in that cell consisting of the habitat woodland. Using the intervention-effect relationships found, information is obtained on the magnitude of the anticipated effect for each aspect of habitat fragmentation and each habitat. For each stretch of road an indication is provided of the magnitude of the impact of fragmentation, by habitat category and by aspect.

This information can be represented cartographically. We have opted to do this on the basis of quartiles (see also: Fig. 2). First the effect scores for each road stretch are noted in ascending sequence. Then they are grouped into quartiles. Each quartile represents 25% of the overall effect. In the first quartile there are a relatively large number of stretches, with each stretch contributing little. In the fourth quartile the opposite holds: here there are relatively few stretches of road, each of which makes a major contribution. The second and third quartiles have an intermediate character.

Figure 2.
Schematic representation of the four quartiles for one aspect of habitat fragmentation, built up from the effects caused by all the road stretches studied in a certain area.

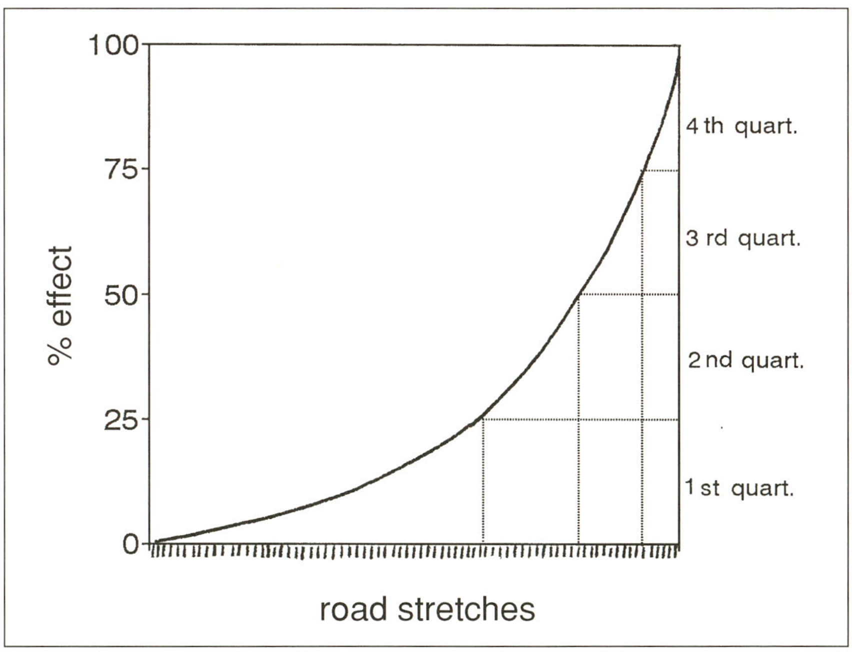


Table 2.
Effect distances for the black-tailed godwit in open meadows for a 120 km/h speed limit and relative decrease in density in a 1000-m zone from the road.
The decrease in density between the road and the effect distance, is approx. 50%, independently of traffic intensity (modified from: Reijnen & Foppen 1991).

traffic intensity (vehicles/day)	effect distance (m)	density decrease in 1000-m zone (%)
<5,000	230	11
5-10,000	360	17
10-25,000	640	30
25-50,000	930	44
>50,000	1,130	48

The various phases of data processing can be illustrated with reference to the black-tailed godwit *Limosa limosa* (habitat category: Grassland) and the sensitivity of this species to disturbance by traffic. Table 2 shows, for various levels of traffic intensity, the effect distance and percentage decrease in density. The decrease in density between the road and the effect distance is - under the circumstances investigated - always approx. 50%, independent of traffic intensity.

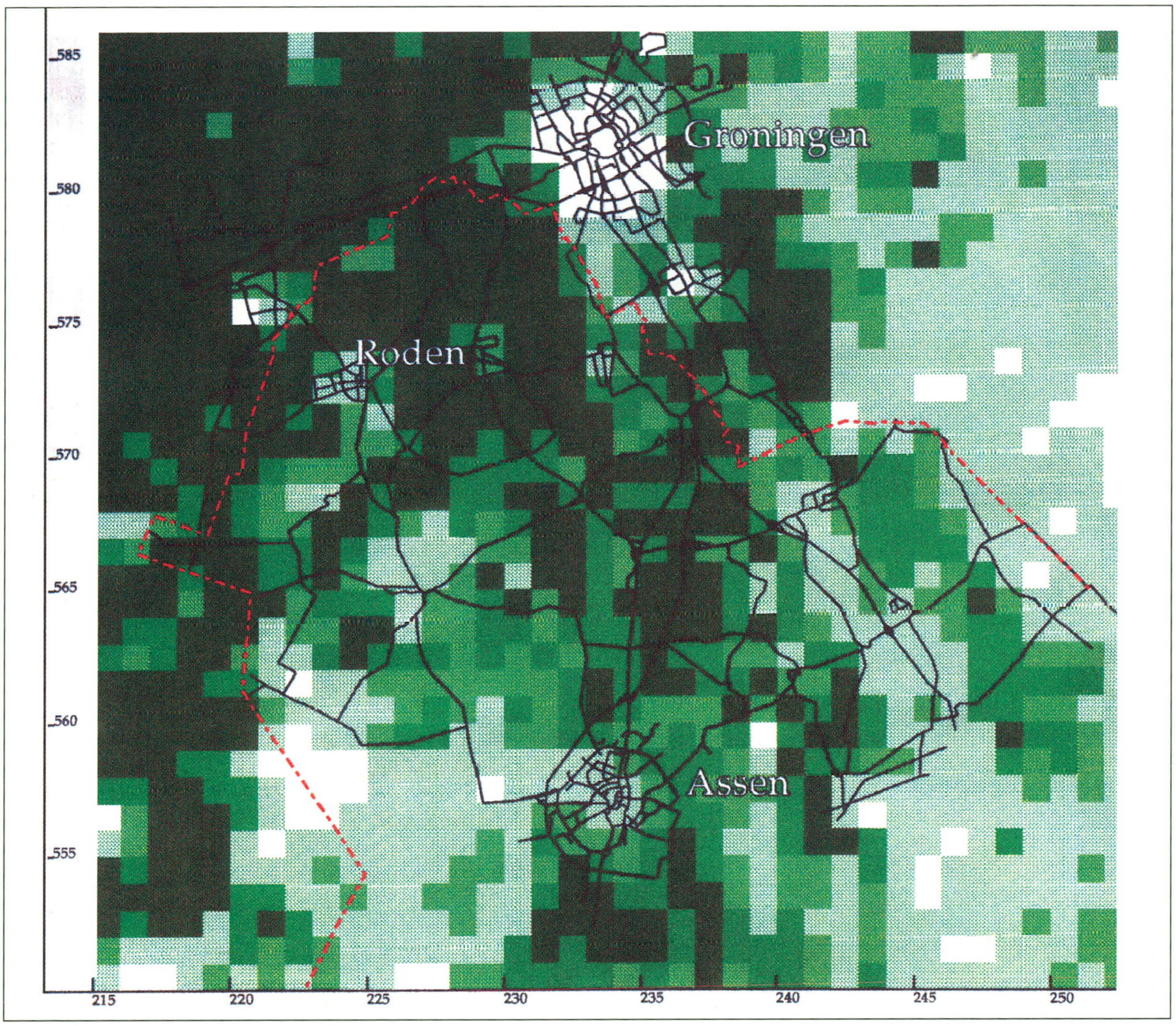


Figure 3a.
Occurrence of the ecotope grassland in the Groningen transport region
one grid cell = 1 km^2;
grassland:
= > 0-25 ha//
= 25-50 ha//
= 50-75 ha//
= 75-100 ha
--- = roads;
- - = provincial boundary.

Results

Figure 3a shows the occurrence of the ecotope grassland in each kilometre grid cell.
Figure 3b gives an indication of potential habitat available for the black-tailed godwit, i.e. Grassland after correction for the presence of built-up areas, woodland and 'linear green elements'.
Figure 3c shows the disturbed area quartiles, i.e. the zones along the roads

in the transport region in which the black-tailed godwit should have been declined in density by approx. 50%, when following the calculation method here proposed. In the case of road stretches from the first quartile the disturbance zone concerned has a width of several hundred metres; for stretches belonging to the fourth quartile the affected area has a width of approx. 1100 metres (see: Table 2).

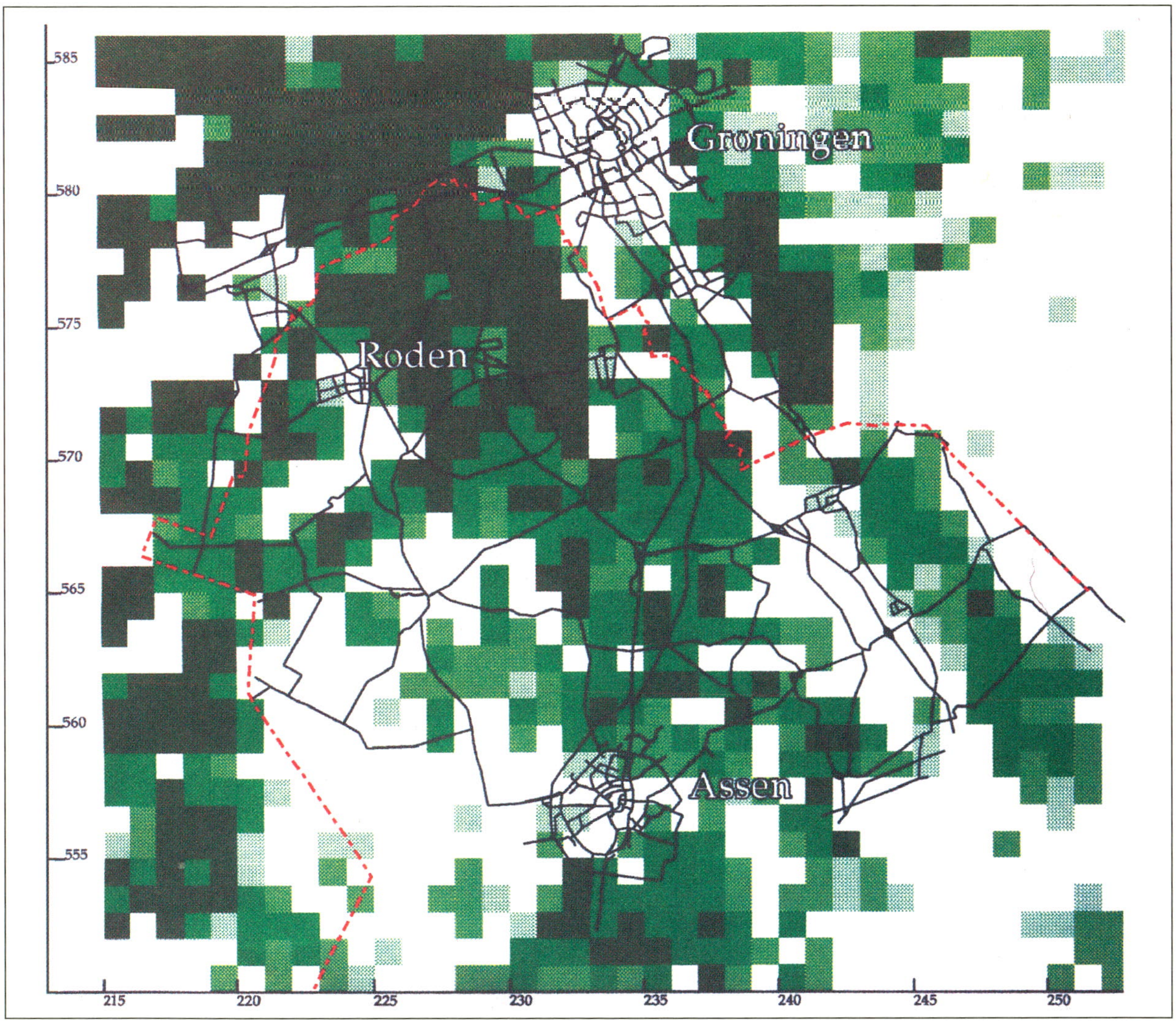


Figure 3b.
Occurrence of the habitat Grassland, i.e. the potential habitat for the black-tailed godwit *Limosa limosa*, in the Groningen transport region (for symbols, see Fig. 3a).

Figure 4 gives an impression of the number of casualties occurring along arable fields. This is based on

1. the number of traffic casualties found by Preesman (1978) for the hare *Lepus europaeus* along a motorway in the west of the Netherlands (5.0/km*y),
2. assumption of a linear relationship between the number of casualties and the speed limit in force on the road - and of course on the assumption of an even distribution of the hare in the habitat or of the suitability for the hare in this category.

The assumption of a linear relationship is based on the fact that lowering the maximum speed from 120 to 100 km/h (a factor 0.8) resulted in a decrease in the number of casualties by a factor 0.7 (Preesman 1978). Because of a lack of data, it was not possible to make calculations using the parameter of traffic intensity.

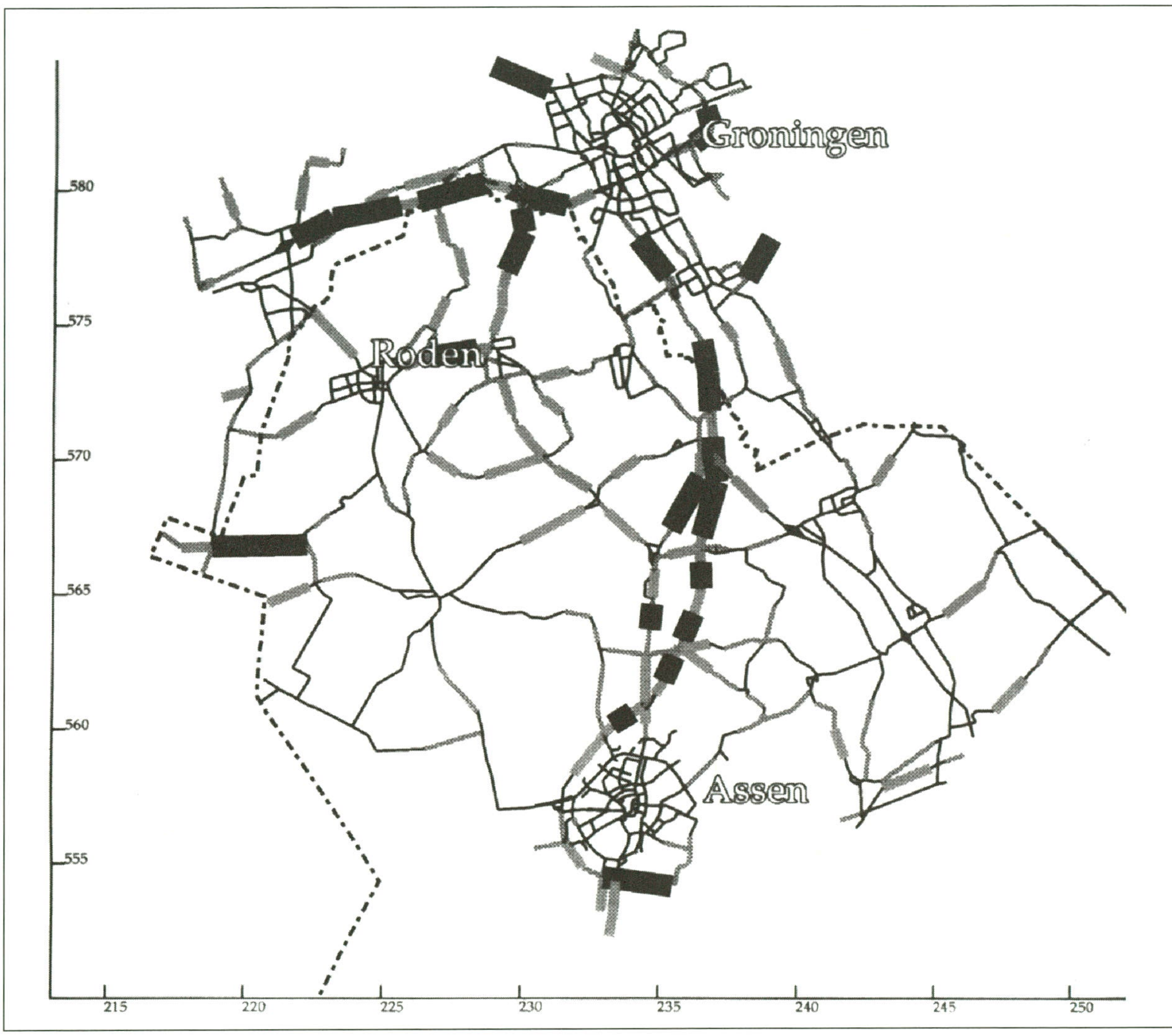


Figure 3c.
Disturbance quartiles for the black-tailed godwit *Limosa limosa* (habitat category: Grassland); the narrowest zones, i.e. the slightest disturbance, belonging to the first quartile, the widest to the fourth quartile (for symbols, see Fig. 3a).

Discussion and conclusions

Because of the lack of comprehensive, detailed data on the actual occurrence of birds and mammals, we were obliged to proceed from the potential presence of selected species. This means that we employed a worst-case approach: since actual occurrence is unknown, we work with the potential occurrence of the species in question. This approach can be justified because the actual presence of the species is partly caused by the actual presence of roads and traffic. Use of the most sensitive species also represents a worst-case approach, although it should be noted that the most sensitive species are by no means exceptions: all the species investigated were found to demonstrate an effect. Moreover, it should be born in mind that we have proceeded from the current state of (empirical) science. Species that already underwent serious decline or local extinction in previous decades - such as the ruff *Philomachus pugnax* and common snipe *Gallinago gallinago*

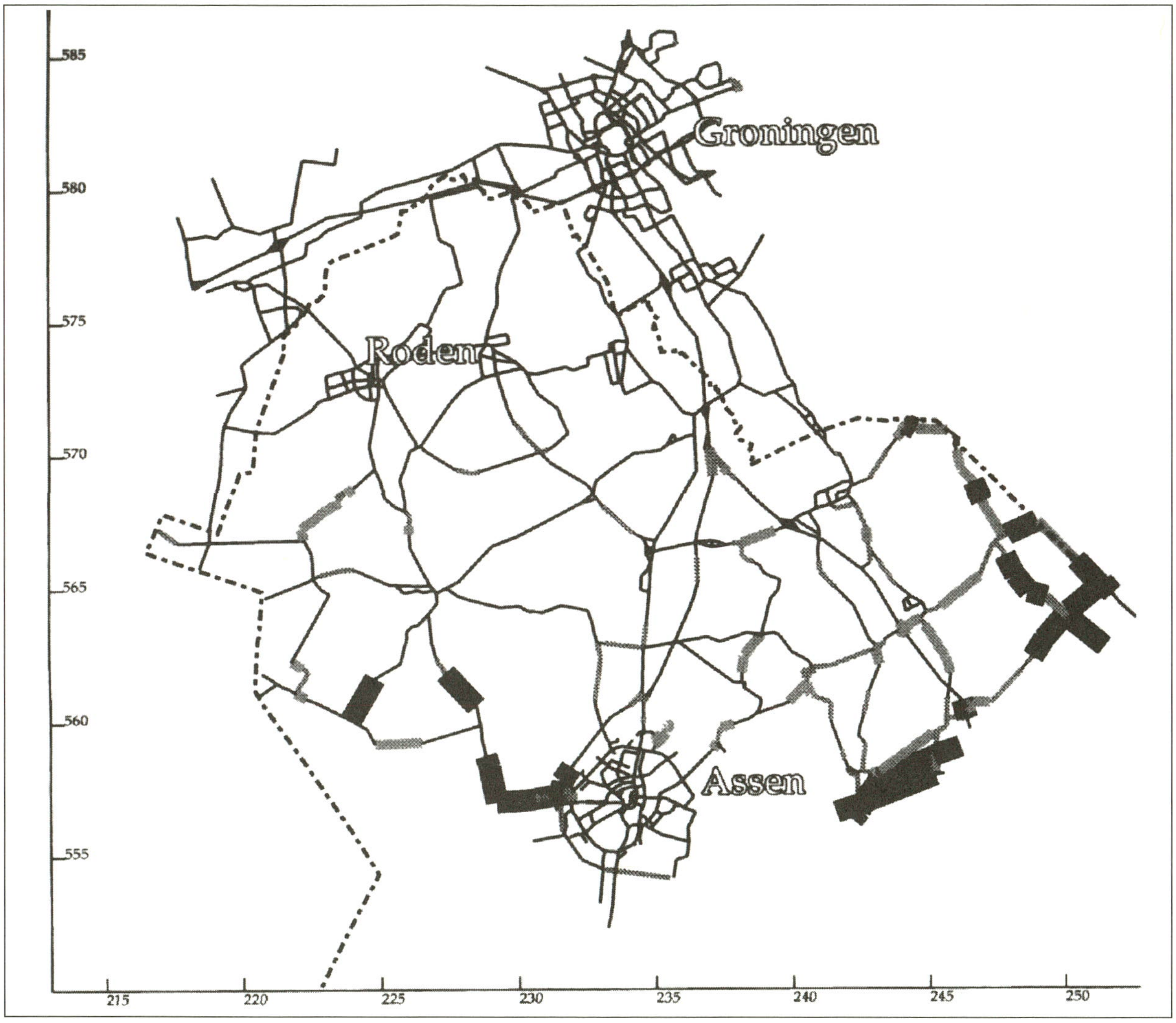


Figure 4.
Collision quartiles for the hare *Lepus europaeus* (habitat category: Arable farmland); the narrowest zones, i.e. the fewest casualties, belonging to the first quartile; the widest to the fourth quartile (for symbols, see Fig. 3a).

in the case of meadow birds - were perhaps even more sensitive. By using data on groundwater level and (other) soil conditions an even more detailed picture can be obtained about the presence and even abundance of the species concerned, i.e. following a Habitat Suitability Index approach.

The relationship between the presence of a habitat and the occurrence of a species deserves comment. At the moment presumed occurrence is based on a rough habitat characterization, with no allowance being made for differences in habitat quality. In the future this might be overcome by including more habitat factors, such as water table, nutrient level, and more

information on vegetation structure and/or maturity. Further study could then indicate whether the 'added value' thus achieved weighs up against the far more complex calculations implied by such an approach; complex, among others, in view of the demands set on the results by users of the evaluation model.
More data are needed about the influence of intensity and speed of traffic on collisions, to produce more reliable results for this aspect of fragmentation.

It was not possible to develop a method for barrier action. The first reason for this was the lack of spatial information on the exact location of the ecotopes - and thus about the location of the habitat categories -, without which there is too much speculation about the intermingling and interference with other habitat categories in the vicinity of a given road. Secondly, in EVV it is still impossible to process such detailed information about these ecotopes, i.e. about polygons. Thirdly, massive database manipulation is implied, which not only requires extensive programming but which also imposes restrictions on the users of the evaluation model.
In anticipation of these obstacles eventually being removed, we have developed three alternative options which can subsequently be examined for their practicability. In the first approach, the 'overlay method', information on the intersection of major ecological connective structures by roads is used. This approach presumes a knowledge of the presence of ecological connective structures and, partly for this reason, implies the inclusion of policy (i.e. normative) aspects in determining effects from an intervention.

In the 'wave propagation' approach, a road is viewed as a 'resistance' that an individual animal must overcome while on the move for the purpose of dispersion, foraging and so on. For a single individual, proceeding from a point of departure x,y, a simulation model can be used to determine the probability distribution of distances traversed in various directions (cf. Knaapen et al. 1992).

The third approach, 'neighbourhood analysis', proceeds from the kilometre grid cell information in the LKN database, in tandem with the information on the 8 or 24 adjacent kilometre cells (one or two rings deep, respectively). In this way the traversibility of the neighbouring cells for a species is calculated incorporating the presence of roads. Manipulation, processing, calculation and presentation of the results would be carried out at the level of road stretches by means of proportional allocation of the kilometre grid cell information to the road stretches, giving many uncertainties about the realistic value of the outcomes when de LKN database is used.
Ranking the quantified effects is a matter of both ecology and policy: how severe is the effect for the species or the ecosystem studied, and what is the nature value given to this species or this ecosytem? By combining the two answers to these questions, effects can be ranked. The next step - choosing possible measures - is governed not only by this initial ranking but also by considerations of cost and (relative) effectiveness. Ultimately, nature conservation measures can be developed and implemented within the context of a regional transport plan.

It has proved possible to develop a method for determining the fragmentation effects 'habitat loss', 'disturbance' and 'collisions' due to roads and traffic in a transport region. A more detailed database with information about the exact location of the ecotopes and possibilities for handling this information in the EVV model will provide a good starting point for also accounting for the 'barrier action' of roads. Research at our institute on this item will be started soon.

The method described here is now implemented in the EVV model (version 2.0) and available for use, i.e. research into the effects of regional traffic and road planning, as the prior versions have allready been used.

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Habitat fragmentation, the role of minor rural roads and their traversability

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Abstract

This paper deals with a systematic way to acquire knowledge about the probabilities of successful road crossing by species and what characteristics affect the traversability. A model is discussed to investigate habitat fragmentation and the role of Minor Rural Roads (MRRs). This model is developed as a tool to predict the effects of mitigating habitat fragmentation of MRRs and their traffic. It is applied in a case that addresses both traffic problems and habitat fragmentation by an integrated regional planning approach.

Introduction

Infrastructure and its traffic flows are principal causes of habitat fragmentation. Defragmentation, as mitigation of habitat fragmentation to facilitate species movement, is an important topic in nature conservation. In the Netherlands, a lot of attention is given to fragmentation effects of motorways and other major roads, especially for locations where they intersect the National Ecological Network, and the way these effects can be mitigated (e.g. van der Fluit et al. 1990). However, these high quality trunk roads represent only 10 to 20% of the overall stock (OECD 1986).
The remainder of the network is here referred to as Minor Rural Roads (MRRs). MRRs are located outside built-up areas and are managed by municipalities. It is apparent that the MRRs have an important impact on plant and animal species and their habitat. Yet, there is a lack of knowledge concerning the fragmentation effects of MRRs. This paper focuses on MRRs in the Netherlands and their role in habitat fragmentation, especially as barriers for species movement forced by both the presence of MRRs and the traffic using MRRs. In this context, a model for traversability is elaborated. Traversability is defined as the probability of successful road crossing by a species.
The objective of this paper is to investigate the relationships between road design, traffic characteristics and traversability of MRRs for selected species. Two questions will be addressed:

1. To what extent will the traversability of MRRs be affected by:
   - road characteristics, such as width of pavement, landscape types along roads,
   - traffic characteristics, such as volumes, time split, speeds, and
   - species characteristics, such as dispersal behaviour, home range, habitat?

2. Can a planning approach based on the spatial concept 'Traffic Calmed Area', as advocated by Jaarsma (1991, 1996), address environmental impacts of MRRs? Can the model be used to predict the effects of changes in road and traffic characteristics?

MRRs and habitat fragmentation

MRRs as part of the road network

The traffic volume of MRRs varies from a few hundred to a few thousand vehicles per day (OECD 1986; Jaarsma 1994). In the Netherlands, approximately 48,500 km of MRRs exist. The length of motorways and rural major roads is 2,000 and 5,500 km respectively. Based on their function, MRRs can be classified into three types:

1. access roads to a few parcels or farms only,
2. access roads with a slight traffic function, and
3. roads connecting villages and rural roads of lower categories.

MRRs of type 1 and 2 have (mainly) an access function, MRRs of type 3 (mainly) a traffic function. These different functions imply different traffic volumes, as shown in Figure 1a.

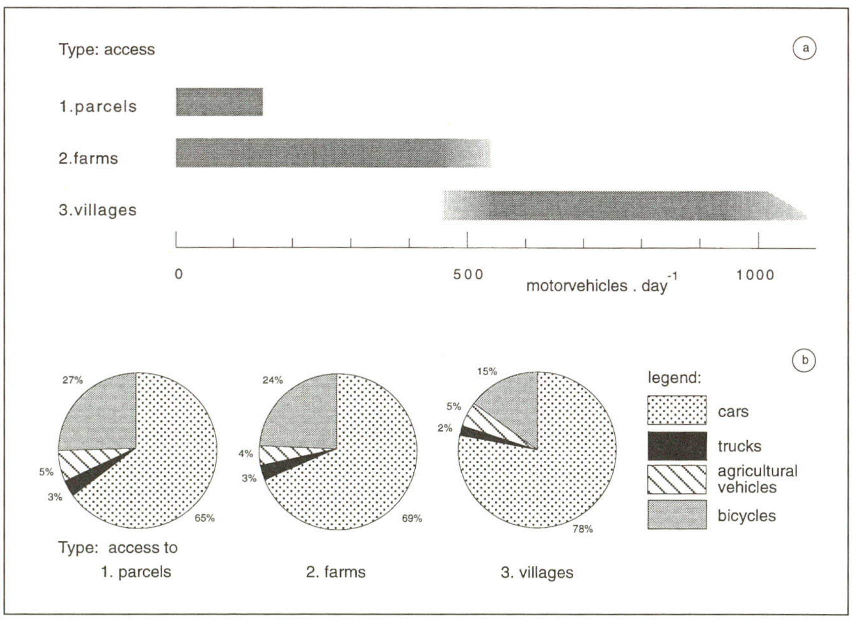

Figure 1.
Minor rural roads: average annual daily volumes (a) and modal split (b) per type (Jaarsma 1994).

An indication of differences in traffic composition on the three categories of MRRs is presented in Figure 1b. It illustrates an important characteristic of MRRs: the mixed composition of traffic by mode. Heavy and light vehicles, slow and fast vehicles, cars and bicycles occur together.
As far as numbers go, passenger cars are very dominant (Jaarsma 1991). The mixed traffic composition results in large differences in speeds: from less than 5 km.h^{-1} for pedestrians, to 15 to 30 km.h^{-1} for cyclists and agricultural vehicles, to 60 to 100 km.h^{-1} for cars (where 80 km.h^{-1} is the legal limit). MRRs are modest in size: most MRRs are one lane roads with restricted pavement width (between 2.5 and 5.5 m).

Due to the traffic and road characteristics and the increasing traffic volumes, several problems appear on MRRs: traffic unsafety, congestion, rat run traffic, annoyance for residents along the road, damage to roads and/or verges and environmental impacts (Jaarsma 1994). These problems arise when the actual use of the infrastructure (more and more) is inconsistent with its function, by which the traffic volume exceeds the capacity of the road. In this context, Jaarsma (1994) emphasizes an increasing discrepancy between function and design of MRRs. This takes place where through-traffic uses MRRs of type 1 and 2, and, to a lesser extent, of type 3. Also, environmental impacts are related to the traffic and road characteristics and the trends in traffic volume.

Adverse effects of MRRs on animal species

Four adverse effects of roads on species can be considered (after van der Fluit et al. 1990):

1. destruction or alteration of habitat due to construction works
2. disturbance of habitat along the roads (noise, vibrations, car visibility, etc.)
3. physical barriers created by roads (increased resistance for movements)
4. barriers by traffic (collision chance during crossing).

The first two effects directly influence the habitat of species. They result in diminishing area of habitat or in strips along roads with marginal habitat for species. Since MRRs already have a dense existing network, new construction is an exception nowadays. In the Netherlands, the meshes of the network are on average 1.5 x 1.5 km (Jaarsma & Monderman 1995).
So far as the disturbance of habitat is related to the noise load (e.g. Reijnen et al. 1995), the distance of disturbance is related to the traffic volumes. So, we expect that the disturbance of habitat along MRRs is less severe than along trunk roads. Therefore, this paper focuses on the barrier effects of both MRRs and their traffic.

The barrier effects of road and traffic affect populations of insects (Vermeulen 1994), reptiles and amphibians (e.g. Vos & Chardon 1994), breeding birds (Reijnen et al. 1995) and mammals (e.g. Mader 1984; Lankester et al. 1991). Due to the barrier effects of roads and traffic, the species will either decide not to traverse, follow another route, or undergo the risk of a collision. The mortality effect of roads is the number of individuals killed by traffic. The relationships between barrier and mortality effect, and characteristics of roads and traffic, are complex (Verboom 1994). Literature focuses mainly on measuring victims and identifying places of high mortality (Engelberts 1976; Davies et al. 1987; van den Tempel 1993). We look for relationships between these indices and traffic and road characteristics. High traffic volumes cause high noise loads and high chances of collision, since the intervals between the vehicles are small. An increase of volume may lead to such a flow of vehicles that individuals are restrained to cross the road. The wider a road, the more time an animal needs for crossing and the lesser the chance of a successful road crossing. Moreover, wide roads induce higher traffic volumes and speeds.

For the Netherlands, a classification of terrestrial mammals to according their vulnerability for fragmentation by infrastructure was made, based on habitat characteristics, dispersal behaviour and home-range size (Table 1; Brouwer 1993). The habitat characteristics are clustered into three landscape types. Open landscape types consist of extensive agricultural parcels. Half-open types are mosaics of hedgerows, forest patches and agricultural parcels. Closed types are characterized by a forested landscape. It appears that mammal species of closed and half-open landscape types with a large home range and well dispersing behaviour are sensitive to fragmentation (Oxley et al. 1974, Adams & Geis 1983). The clearance of roads, defined as the distance between the forest edges on both sides of the road, is mostly small in closed and half-open landscapes. The roads of these types are mainly MRRs. It appears that species are inclined to traverse these roads. The species of cluster 5, 6, 7 and 8 are relatively vulnerable to fragmentation by infrastructure, due to their large home range, dispersal behaviour (cluster 6, 7 and 8), and preference for closed landscape types (cluster 6 and 8).

Clusters of mammal species	Landscape types[1] open	Landscape types[1] half-open	Landscape types[1] closed	Dispersal behaviour[2]	Home range[3]
1 mole, white-toothed shrew, water shrew, bicoloured shrew, ground vole, wood mouse, harvest mouse	+	+			S
2 pygmy shrew, common shrew	+	+	+		S
3 bank vole, short-tailed vole, pine vole, dormouse, garden dormouse	+	+			S
4 root vole, common vole, black rat, common hamster	+	+			S
5 muskrat, brown rat, stoat, otter		+			L
6 hedgehog, weasel, polecat, beech marten		+	+	D	L
7 brown hare, rabbit, red fox, roe deer	+	+		D	L
8 red squirrel, pine marten, badger, red deer, wild boar		+	+	D	L

[1] + = the landscape type contains the habitat of the species
[2] D = species with relatively extended daily, seasonal or migratory movements
[3] S = species with relatively small home range (< 1 ha), L = species with relatively large home range (» 1 ha)

Table 1.
Classification of terrestrial mammals based on habitat, dispersal behaviour and home-range size (Brouwer 1993).

The role of MRRs in habitat fragmentation is probably substantial. However, few quantitative data are available. Klerks et al. (1994) argue that MRRs contribute to traffic victims of mammals, birds, reptiles and amphibians. First, MRRs are relatively small, so that the presence of the road acts less frequently as a physical barrier for traversing. Second, MRRs generally have a low clearance. Third, MRRs have a relatively low noise load. Fourth, the road density is high and the mesh size of MRRs is small. Berendsen (1986) divides the number of traffic kills of the badger *Meles meles* in the Netherlands over road categories: 46% of the badger collisions concern MRRs. Among barn-owls *Tyto alba* killed by traffic, about 23% are caused by traffic on MRRs (van den Tempel 1993).

Mitigating measures

Mitigation of habitat fragmentation involves technical devices and other measures to reduce the barrier effects of road and traffic. These measures are intended to improve the connectivity between fragmented habitat patches. However, devices such as fauna passages or fences are not the only mitigating measures. Measures such as reduction of traffic volume or speed, or (limited) closing of roads may also reduce the barrier effects. Evaluation of the effectiveness of mitigating measures appears to be very complex. Opdam (1994) enumerates several indices to assess effects at the population level: the survival chances of the (meta)population, differences in occupancy or density of habitat at certain distances from the road and extinction chances in patches near the road. However, these indices depend on various factors: species characteristics and landscape features, such as the spatial configuration of habitat along the road. Indices at the individual level are relatively simple to estimate. Two indices can be distinguished: the actual use of fauna passages and the number of traffic victims. These indices can assess the intended effects of the technical devices. It is also possible to compare the effectiveness of different devices. However, the population effects of a reduction in mortality by traffic are difficult to determine as long as other factors are unknown. Therefore, the indices on individual level are unsuitable for predicting effects of mitigation on population persistence. Exceptions are counted for species whose dominant cause of mortality is traffic. For example, the populations of badgers (Lankester et al. 1991) and barn-owls (van den Tempel 1993) in the Netherlands are largely affected by traffic collisions.

Nevertheless, knowledge concerning effects at the individual level can be used to estimate barrier and mortality effects of infrastructure. This may lead to increasing knowledge of fragmentation effects at the population level. The effects of roads and traffic should be understood by investigation into the extent to which movement is influenced by road, traffic and species characteristics. Nieuwenhuizen & van Apeldoorn (1994) started to evaluate the use of fauna passages by mammals. Yet, conclusions about the effectiveness of mitigating measures are difficult given the lack of comparable data of the number of victims before and after the introduction of mitigating measures.

Methods

Spatial concept 'Traffic Calmed Area'

We advocate an integrated regional planning approach to address both traffic and environmental problems (Jaarsma et al. 1995). This marks a transition to road planning in a wider perspective, as advocated by OECD (1986). It also marks a transition from planning for road links to planning for road networks (Jaarsma 1996). This approach acknowledges the fact that planning interventions affect a larger area than the direct surroundings of the device. This holds for both traffic and species. Moreover, problems at one place may be remedied by interventions at another place.

The spatial concept 'Traffic Calmed Areas' can guide planning interventions at a regional scale (Jaarsma 1991). The concept 'Traffic Calmed Areas' is a spatial organisation principle for infrastructure containing several engineering consequences. A traffic calmed area is a region containing roads with an access function, which is enclosed and accessible by major roads. The concept provides a way for (re)assignment of traffic functions and adaptations of the design resulting in a (re)organisation of traffic flows: diffuse volumes at the MRRs will be concentrated at a few major roads. Traffic volumes and speeds within the traffic calmed area will decrease. It leads to a restricted number of MRRs with a traffic function and related higher volumes, whereas most MRRs only have a local access function with low volumes, restricted speed and (eventually) an adapted design. Forcing back through-traffic from the MRRs to near major roads causes a slight increase of total travel distance, travel time and energy consumption.

Model for traversability of MRRs
A model has been developed to explore the resistance of roads and traffic for species and the number of traffic victims. In the context of the foregoing, it is reasonable to assume that the probability of successful road crossing depends on characteristics of species, of roads and of traffic.
Such a model should be based on the assumption that a road crossing of an animal is successful if an 'acceptable' gap in the traffic flow appears at the start of the crossing. A crossing during a smaller gap results in a collision, since an animal and a vehicle are at the same place at the same moment.

This gap acceptance approach has been worked out for waiting times for pedestrians as a function of traffic volumes (Hunt & Abduljabbar 1993). For the application of this approach to animal species several suppositions should be made (van Eupen & van der Veen 1995). Species will traverse roads without any waiting time, especially in situations with a low clearance. Since the strategies used by species to traverse the road are unknown, it is assumed that they cross in a right angle and with a constant speed. For some species, this is doubtful since they will flee or stay. For most species, the traversing speed will be underestimated due to their (unknown) flee behaviour as soon as a vehicle approaches. It is assumed that the traversing speed for an adult mammal species is 0.25 times its maximum speed (this is comparable with the norm of the traversing speed of a human adult). So, the time C needed for a road crossing can be calculated from:

$$C = \frac{(B + L_i)}{V_i} \qquad (1)$$

where B is the pavement width of the road, L_i is the body size of the species (measured from snout to tail tip) and V_i is the traversing speed of the species. If the road has two lanes, for both lanes $\frac{1}{2}C$ seconds are needed.

In traffic engineering the calculation of the probability of gaps with a certain duration in a traffic flow is commonly based on the assumption of a Poisson distributed process (Drew 1968, Leutzbach 1988). This means that the number of vehicles in a sequence of fixed periods of time is Poisson distributed, the numbers of gaps between vehicles at a certain place are (negatively) exponential distributed, and the length of the time periods of these gaps is independent. So, the theoretical probability P_i of a successful two-lane road crossing for an animal species i can be viewed as the probability of a gap of $\frac{1}{2}C$ in the first lane, directly followed by an equal gap in the second lane. This probability only depends on the volumes. It can be formulated in a decreasing exponential function, written as:

$$P_i = e^{(-\lambda \cdot \frac{1}{2} C)} \qquad (2)$$

where λ is the decisive traffic volume of the two lane road. The decisive volume refers to the volumes at the periods of the day or season that correspond with the species' periods of movement (foraging, dispersal, etc.). Traffic speeds are not explicitly included in this formulation. Speed of vehicles is incorporated in the duration of the time interval between vehicles.

Based on this equation, the number of traffic victims D_i can be estimated by:

$$D_i = (1 - P_i)\, k_{i,t} \qquad (3)$$

where $k_{i,t}$ is the number of road crossings for individuals of species i in the time period t. The parameters V_i and $k_{i,t}$ are ambiguous factors, which may influence the value of P_i and D_i. Therefore, it appears not realistic to use P_i and D_i as predictors of the real traversing probability and numbers of traffic kills. They can be used to estimate the relative change of the traversability ΔT_i due to mitigating measures (van Eupen & van der Veen 1995), written as

$$\Delta T_i = \frac{P_{i,1} - P_{i,2}}{P_{i,opt} - P_{i,1}} \qquad (4)$$

where parameter $P_{i,1}$ is the chance of successful road crossing in the actual situation and $P_{i,2}$ for the planned development with width B_2 and traffic volume λ_2. The relative change of the traversability ΔT_i of both P_i and D_i is calculated with regard to the optimum situation ($P_{i,opt} = 1$ and $D_{i,opt} = 0$). A decrease of λ and/or B will result in an increase of P_i and a decrease of D_i, and, therefore, in an increase of ΔT_i. Due to the species-specific V_i and L_i, the effects of changing B and λ, and ΔT_i differ between species.

For calculating the consequences of spatial planning on traffic flows, a transportation model is used. This method has been implemented in the GIS Arc/Info. The combination of this method and the traversability model provides a tool for calculation of traffic volumes and prediction of effects of habitat (de)fragmentation. This model can be used to assess several traffic scenarios with mitigating measures in rural areas. These scenarios vary in technical design of MRRs and measures for defragmentation and traffic regulation.

Case study

The model has been applied in a case study in the south-eastern part of Friesland, one of the northern provinces of the Netherlands: the forested regions around Ooststellingwerf. This area is part of the forest complex Fries-Drentse Wouden. It is designated as core area in the National Ecological Network (Provincie Friesland 1994). The area Ooststellingwerf with core areas, wet and dry corridor zones is shown in Figure 2. The study area contains several forests and the Tjonger brook.

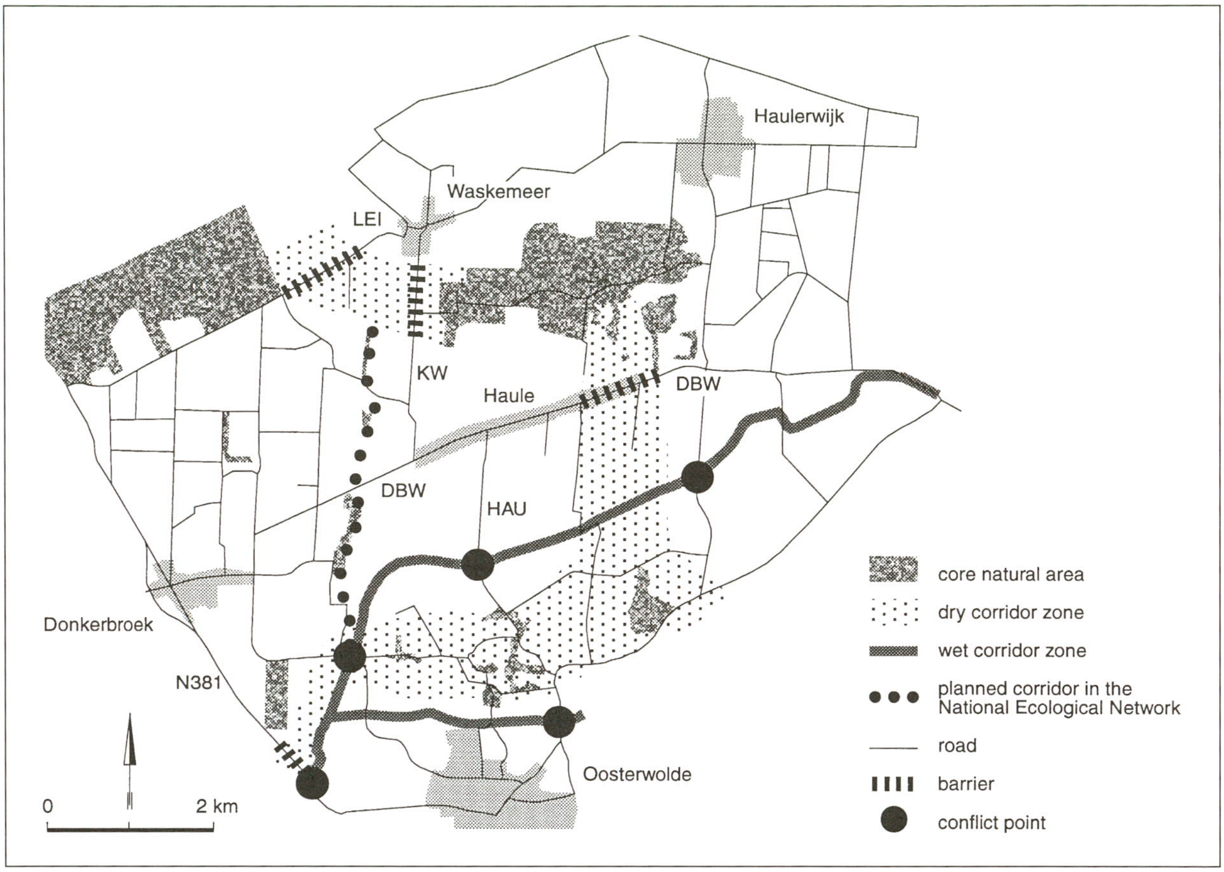

Figure 2.
Ecological network of Ooststellingwerf (Provincie Friesland 1994).

Several MRRs act as barriers in the corridor zones: Kruisweg (KW), Leidijk (LEI), Dorpsstraat and Bovenweg (DBW), Haulerdiek (HAU) and the major road N381. The barriers and local conflict points for species movement are indicated. Especially for larger animals, like roe deer *Capreolus capreolus*, red fox *Vulpes vulpes* and mustelids (pine marten *Martes martes*, beech marten *Martes foina*, stoat *Mustela erminea* and weasel *Mustela nivalis*), the roads act as barriers. A population of pine marten exists in the Fries-Drentse Wouden. The major mortality of pine marten in this area is caused by traffic collisions (G Müskens unpubl. data). Besides, recent policy and nature restoration plans intend to reintroduce the badger and the otter *Lutra lutra* in the Fries-Drentse Wouden. These populations will certainly be affected by the traffic in the study area. The barrier effects are mainly caused by the high traffic volumes and high (legal) speeds. The area is considered to be important for the pine marten as corridor zone between populations of the forests of Beetsterzwaag, Duurswoude and Appelscha (Provincie Friesland 1994). The environmental and traffic problems are discussed by Jaarsma et al. (1995).

The objective of the case is to illustrate changes in traversability and traffic flows due to planning solutions following the concept 'Traffic Calmed Area'. It is worked out for the roe deer (the assumptions are: L_r = 1.4 m and

V_r = 5.2 m.s^{-1}). As shown in Table 1, the species is sensitive for fragmentation due to its home-range size, dispersal behaviour and habitat characteristics. For predicting effects, it may represent other species such as the pine marten. For the roe deer, collisions mostly occur in the twilight and at night. The decisive λ is the volume measured in these periods (estimated as 25 per cent of the average traffic volume per day). Hence, ΔT_i is calculated for these traffic volumes.

Results

The autonomous development of volume within 10 years is calculated (expected trend of increasing volume of 1 per cent.year^{-1}), see Figure 3a. The planned development for a traffic calmed area is presented in Figure 3b. This is one of the scenarios worked out by Jaarsma et al. (1995). In this plan, the N381 and Leidijk are assigned with a traffic function. The remaining roads of interest are classified into the category of access roads. The traffic volumes of these roads will be reduced drastically. Due to lower volumes and smaller pavement width, there is a substantial positive effect on the traversability. An overview of the results of the model is presented in Figure 4. In Figure 4a, the change in volume and width of pavement for the MRRs is indicated. Figure 4b contains the relative change in traversability related to the current situation and to the planned development. For KW, DBW and HAU, a positive change can be expected for the planned development.

The spatial accessibility of the Ooststellingwerf area will not be limited. For some places, the accessibility in time increases somewhat. The travel time for crossing the whole area will increase 10 per cent (Jaarsma et al. 1995).

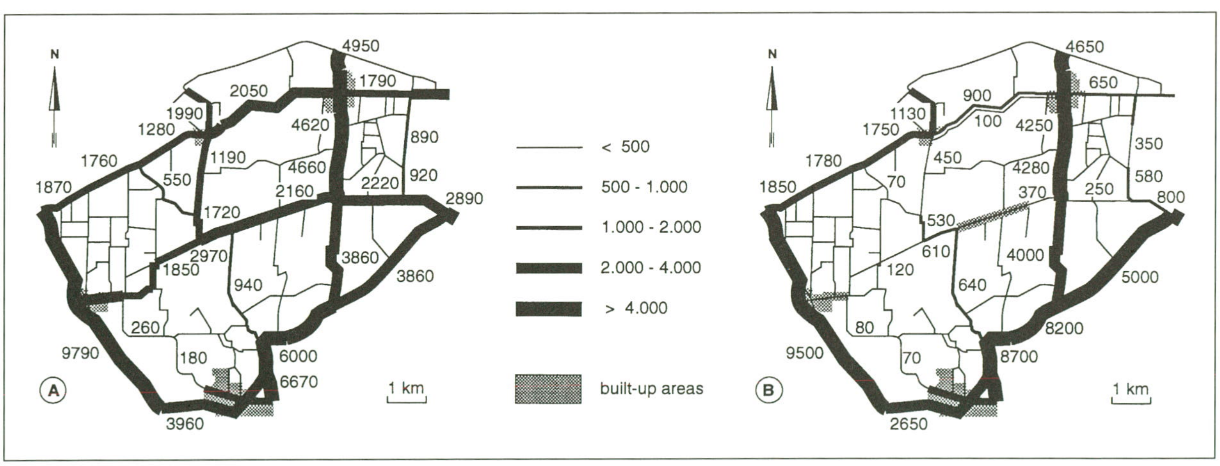

Figure 3.
Autonomous (a) and planned (b) development of traffic volume in vehicles.day^{-1} in Ooststellingwerf (Jaarsma et al. 1995).

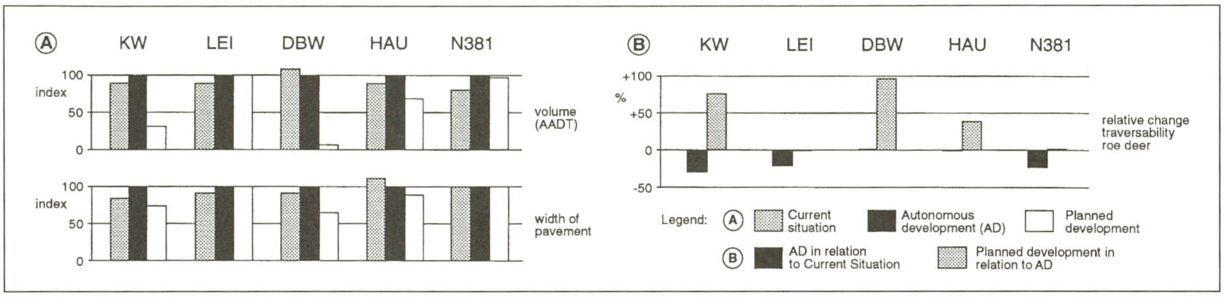

Figure 4.
Changes in road and traffic characteristics and effects for the roe deer in Ooststellingwerf for the actual situation in 1994, autonomous development within 10 years, and the planned development.

Discussion and conclusions

In the model, the theoretical traversing probability is determined by the road characteristic of pavement width, the traffic characteristic of volume, and the species characteristics of traversing speed and body size. The effects of changing volume and pavement width are illustrated in Figure 5. Drastic reduction of volume has the largest effects, especially in combination with narrowing the pavement width. The effects of the species' traversing speed and body size are minor: an exact estimate of V_i and L_i is not necessary. We found a slight difference between ΔT_i and P_i for roe deer and pine marten (L_m = 0.7 m and V_m = 6.4 m.s^{-1}). It appears that ΔT_i and P_i are strongly affected by V_i if $V_i < 2$ km.h^{-1} and λ is unlikely high (van Eupen & van der Veen 1995). Besides, several other factors may be important which are not included in the discussed model: traffic speed, noise level of approaching vehicles, direction of road crossing, individual movement or in groups, and the clearance of the road. Application of the model in the case area showed that the theoretical traversing probability might be useful for effect prognosis. However, further research should be done to validate the model in practice and to investigate the remaining explanatory factors. Especially, the role of traffic speed needs further research: the relationships between speed and correction possibilities of the driver, flee behaviour of species and turbulence in the proximity of vehicles should be included.

The concept 'Traffic Calmed Areas' has substantial implications for the regional traffic processes. From several points of view, traffic concentration on a limited number of roads with a suitable design is preferable to diffuse traffic on the MRR network. The roads with a traffic function are clearly distinguished. The remaining roads with an access function have relatively modest volumes. The case shows that reduction of traffic volume in combination with smaller pavement width will substantially contribute to defragmentation of habitat by MRRs. This effect will be strengthened by adapted pavement types for MRRs with an access function, for example partially paved types (Unger 1986). Such roads have low barrier effect for small mammals. They will also lead to a decrease of the mortality effect (related to low volumes and speeds on such roads). A relevant question emerges: what area should be covered by a traffic calmed area to contain a minimum viable population of certain species? Besides defragmentation effects, traffic safety and opportunities for outdoor recreation will be enhanced, while the spatial accessibility of the traffic calmed area is guaranteed (Jaarsma 1994).

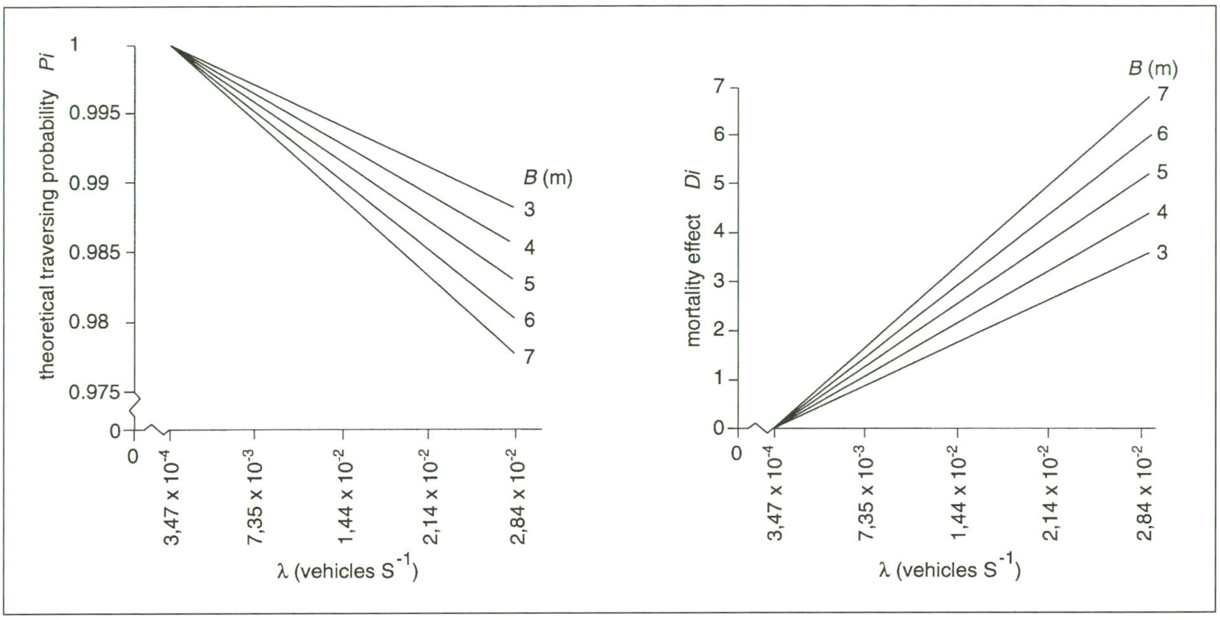

Figure 5.
Theoretical traversing probability P_i and mortality effect D_i for the roe deer based on varying pavement width B and volume λ ($k_{i,t}$ = 300 as the number of road crossings per time period t for individuals of species i).

In spatial planning, attention should be given to assessing the effectiveness of mitigating measures. It is a task for (applied) landscape ecology and spatial planning to provide knowledge and tools for quantitative evaluation of interventions. For both disciplines, the challenge is to acquire knowledge and to uncover uncertainties. In this context, the model discussed in this paper can be a useful tool.

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