A quest for the role of habitat quality in nature conservation
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6 Effects of heavy metals on the badger *Meles meles*; interaction between habitat quality and fragmentation

Abstract

Many wildlife species are threatened with extinction caused by destruction and degradation of their habitat. Habitat size reduction results from reclamation of nature by man, whereas habitat quality reduction on the other hand results from pollution, changes in abiotic conditions such as nutrients, biotic conditions like reduction in prey species, and changes in the structure of the habitat (e.g. depletion of nesting territories). Reduction in habitat quality by pollutants like heavy metals can decrease the survival and reproduction of individuals, and in this way increase the extinction probability of populations. However, it is unclear to what extent reduction in habitat quality caused by pollution reveals itself at the population or community level.

During the last century the badger has shown a strong decline in The Netherlands. Conservation of the species has been mainly directed at reducing traffic accidents, the major cause of death for badgers in The Netherlands. This high mortality, up to 20% a year, has been interpreted as primarily resulting from habitat fragmentation. However, deterioration in habitat quality may also be contributing to the traffic mortality.

In this paper the effect of a decline in habitat quality resulting from cadmium (Cd) and or copper (Cu) pollution of the soil is considered. Earthworms, the major food source of badgers, readily accumulate Cd from the soil without adverse effects at Cd levels occurring in Dutch soils. On the other hand, population growth in earthworms is retarded in soils polluted with Cu although Cu accumulation is minimal. By eating earthworms badgers may thus suffer from kidney lesion, a direct effect of Cd, or experience food scarcity, an indirect effect of Cu. These risks for badgers in The Netherlands were studied. The probability of kidney lesion was assessed by a bioaccumulation calculation and a population dynamics model was used to study the probability of food scarcity. The results show that both habitat fragmentation and reduced habitat quality can play a role in the high traffic mortality.
Introduction

In densely populated areas like The Netherlands the natural environment is reduced to small areas, surrounded by cultivated land and cities, and segregated into patches by infrastructure. These patches may be too small to sustain viable wildlife populations. In particular if these patches are isolated from each other the survival probability of populations can be low, since especially small isolated populations run a high risk of extinction due to demographic and environmental stochasticity (Goodman 1987; Gilpin & Hanski 1991).

In The Netherlands, a political initiative has been put forward to counter the high extinction risk of small isolated populations. This ‘National Ecological Network’ integrates patches of natural habitat into a large network by connecting them with corridors. Furthermore, the amount of natural habitat is increased by reclamation of cultivated lands. In this way, local extinction can be countered by colonisation, the so-called rescue effect. However, it is not evident whether these improvements in habitat size, by increase in the amount of habitat and its connectivity, are sufficient to ensure sustainable populations. It seems plausible that both habitat size and quality determine the survival probability of populations. Habitat quality may play an even more important role than habitat size since an increase in habitat quality has a larger impact on the survival probability of a population that runs the risk to go extinct due to demographic stochasticity than a comparable increase in habitat size (Klok & De Roos 1998).

Pollutants like heavy metals reduce habitat quality. These substances can decrease the survival of populations both directly and indirectly. Direct effects result from intoxication of the species. An example of a direct effect is the decline in the American robin *Turdus migratorius* resulting from secondary poisoning with DDT (Wallace et al. 1961). Indirect effects on the other hand result, for example, when a pollutant decreases the survival of prey populations. The decline in the grey partridge *Perdix perdix* has been interpreted as an indirect effect of herbicide use (Potts 1986; Potts & Aebischer 1995). This decline mainly followed from the low survival of young partridges, which in turn resulted from a lack of insect food. Due to herbicide spraying weed plants relevant for these insects were virtually absent, leading to low insect levels in cereal fields.

The badger *Meles meles* is an example of a mammal species that has shown a strong decline in number and distribution in The Netherlands over the last century (Fig. 6.1). The decline was extreme in the period 1960-1983 when densities dropped with 36% (Wiertz 1993). The cause of the decline has been attributed to hunting and poisoning, destruction of badger habitat and nesting sites (setts), and traffic mortality (Van Wijngaarden & Van de Peppel 1964; Wiertz & Vink 1986). Since 1947 Dutch law prohibits hunting of badgers in The Netherlands, and the badger is fully protected
under the Nature Conservancy Act since 1993. Recent conservation actions intended to increase the population survival of badgers have been directed to reduce the traffic mortality. This mortality has been attributed to the high density of infrastructure throughout the range of the badger (Van der Zee et al. 1992). Therefore, effects of habitat fragmentation have received much interest (Lankester et al. 1991). The conservation actions mentioned above have increased the number of badgers over the last decades (Wiertz 1993; Van Moll 1997). However, in the range of badger densities found in western Europe (Table 6.1) densities in The Netherlands are still relatively low. A possible explanation is that in The Netherlands badger densities may be limited by habitat quality. Pollution can have a negative impact on habitat quality. Effects of habitat quality reductions due to pollution on badgers have received little attention in literature (but see Ma & Broekhuizen 1989).

In this paper the effects of cadmium and copper on the badger were investigated. Cd has a direct effect on the badger resulting from secondary poisoning and Cu inflicts an indirect effect on the badger through food shortage. The risk of secondary poisoning
Table 6.1  
Badger densities in western Europe, after Mitchell-Jones et al. 1999.

<table>
<thead>
<tr>
<th>number km²</th>
<th>region</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.1-0.6</td>
<td>Czechoslovakia</td>
</tr>
<tr>
<td>0.7</td>
<td>Poland</td>
</tr>
<tr>
<td>1.0</td>
<td>The Netherlands</td>
</tr>
<tr>
<td>0.5-1.6</td>
<td>France</td>
</tr>
<tr>
<td>2.4-3.2</td>
<td>Sweden</td>
</tr>
<tr>
<td>2.0-4.0</td>
<td>East Germany</td>
</tr>
<tr>
<td>1.1-6.2</td>
<td>Scotland</td>
</tr>
<tr>
<td>4.7-19.7</td>
<td>England</td>
</tr>
</tbody>
</table>

Table 6.2  
Earthworms in the diet of badgers.

<table>
<thead>
<tr>
<th>volume percentage</th>
<th>region</th>
<th>source</th>
</tr>
</thead>
<tbody>
<tr>
<td>30</td>
<td>Great Britain</td>
<td>Creswell &amp; Harris 1988</td>
</tr>
<tr>
<td>16</td>
<td>Great Britain (Essex)</td>
<td>Skinner &amp; Skinner 1988</td>
</tr>
<tr>
<td>50</td>
<td>Great Britain (Scotland)</td>
<td>Kruuk &amp; Parish 1982</td>
</tr>
<tr>
<td>31</td>
<td>Switzerland</td>
<td>Stocker &amp; Lüps 1984</td>
</tr>
<tr>
<td>47</td>
<td>The Netherlands</td>
<td>Wansink 1995</td>
</tr>
<tr>
<td></td>
<td>(Northwest Utrecht)</td>
<td></td>
</tr>
<tr>
<td>19</td>
<td>Denmark</td>
<td>Andersen 1954</td>
</tr>
</tbody>
</table>

was assessed by calculating the bioaccumulation of Cd, whereas a population dynamics model has been used to estimate the level of food shortage.

Effects of cadmium and copper on the badger

The badger is an omnivore feeding on earthworms, insects, snails, rabbits, grasses, maize, nuts and seeds (Wiertz 1976; Neal 1986). As is indicated in Table 6.2 earthworms can make up a substantial proportion of the diet. Especially *Lumbricus terrestris* and *L. rubellus* are frequently consumed. Earthworms entail a high risk on secondary poisoning for badgers. By feeding on earthworms in areas polluted with heavy metals badgers can take up large quantities of heavy metals since earthworms accumulate them to a great extent. Cd for example can be concentrated in earthworms up from 20 to a 100 times the soil concentration (Ma 1983a). In contrast to Cd, Cu is virtually not accumulated by earthworms (Ma 1983a), and so does not lead to
secondary poisoning. Apart from badgers earthworms themselves may suffer from intoxication by heavy metals. Laboratory bioassays indicate that Cu is lethal to earthworms living in sandy soils if the soil concentration is above 150 mg kg\(^{-1}\) soil, whereas sublethal effects occur when Cd or Cu levels are above 60 mg kg\(^{-1}\) soil (Ma 1983b, 1988; see Table 6.3). Sublethal effects in earthworms, like retarded individual growth and decreased cocoon production, lead to a decrease in population growth rate (Klok & De Roos 1996; Klok et al. 1997) and expected population density (Bavec & De Roos 1996). Badgers living in areas with a high Cd or Cu load may therefore be confronted with decreased availability of earthworms.

Table 6.3 shows the range of Cd and Cu levels detected in Dutch soils. Given the range of Cd concentrations (0.1 - 5.7 mg kg\(^{-1}\) with 95% below 1.0 mg kg\(^{-1}\)), the capacity of earthworms to concentrate Cd to a great extent, and the importance of earthworms in the diet of badgers, Cd may lead to secondary poisoning in badgers. On the other hand food shortage is not expected since in this range of Cd levels adverse effects on earthworms are not very likely. Literature data indicate that sublethal effects in earthworms are expected for Cd concentrations above 60 mg kg\(^{-1}\) soil (Table 6.3). Contrary to cadmium, copper levels in Dutch soils (1.0 -133 mg kg\(^{-1}\) with 95% below 25 mg kg\(^{-1}\)) may in some areas lead to low earthworm densities, because adverse effects on earthworms are expected for Cu soil concentrations above 60 mg kg\(^{-1}\) soil (Table 6.3).

**Kidney lesion as a direct effect of Cd on the badger**

Kidney lesion is one of the first symptoms of Cd intoxication in mammals and birds (Scheuhammer 1987). The age at which badgers may suffer from kidney lesion will depend on the daily intake of Cd, the proportion of the ingested Cd in the gut taken up into the bloodstream, and the distribution of Cd over the internal organs.

The daily intake of Cd depends on the diet composition, the Cd contents of the
food items and the quantity of these food items eaten. This paper concentrates on the risk of secondary poisoning resulting from feeding on earthworms because earthworms accumulate Cd to a much greater extent than other food items consumed by badgers, and furthermore earthworms make up a high proportion of the diet (Table 6.2). The accumulation of Cd by earthworms is, next to the Cd concentration in the soil, very sensitive to the pH of the soil (Ma et al. 2000). Equation 6.1 relates the Cd concentration in earthworms and soil, as has been found by Ma et al. (2000) on the basis of literature data referring to field studies with earthworm species, Cd concentrations, and pH values relevant for Dutch soils.

\[
\log 10[Cd]_{\text{earthworm}} = 2.60 + 0.49 \cdot \log 10[Cd]_{\text{soil}} - 0.20 \cdot \text{pH} \quad (6.1)
\]

Adult badgers with a live weight of around 10 kg consume about 1 kg food per day (Kruuk 1978). The proportion of earthworms in this daily food intake can be fixed at 30% (roughly the mean in Table 6.2).

Which part of the ingested Cd in badgers is transported to the bloodstream is unknown. Bird and mammal studies reveal that the largest part of the ingested Cd passes the gut. Only a small proportion is taken up into the bloodstream, for mammals this proportion amounts to 0.5% (Engström & Nordberg 1979; Lehman & Klaasen 1986). The Cd is distributed mainly over the kidneys and liver, in dependence of the Cd dose. At low doses (< 0.5 mg kg\(^{-1}\) wet-weight day\(^{-1}\)) the kidneys accumulate two to three times as much Cd as the liver (Scheuhammer 1987). Furthermore, kidney lesion in mammals is expected if the kidneys contain more than 200 \(\mu g\) Cd g\(^{-1}\) kidney tissue in dry-weight (Nicholson et al. 1983; Ma & Broekhuizen 1989). At higher Cd concentrations the efficiency of the filtration function of the kidneys decreases, and Cd is found in urine (Goyer et al. 1984). This leads to a change in the Cd distribution over kidneys and liver. Based on the results of studies in birds and mammals referred to above, it is assumed that until the critical Cd load is reached (200 \(\mu g\) Cd g\(^{-1}\)) Cd is distributed in a ratio 2:1 over kidneys and liver. Furthermore, the assumption is made that below this critical load neither active nor passive elimination of Cd takes place.

Given a daily Cd intake equal to \(DI_{Cd}\) \(\mu g\) per day, of which a fraction \(U_B\) is absorbed in the bloodstream the time \(t\) (in years) it takes badgers to reach the critical Cd kidney concentration \(C_L\) is given by:

\[
t = \frac{C_L \cdot (O_K + O_L)}{2 \cdot DI_{Cd} \cdot U_B \cdot 365} \quad (6.2)
\]

where \(O_K\) and \(O_L\) are the organ mass of kidneys and liver, respectively.
In the calculations the mean kidney weight of adult badgers $O_K$ is fixed at 32.5 g (range 20-45 g) and the mean liver weight $O_L$ at 355 g (range 270-630 g; Müskens & Broekhuizen unpublished results). $DI_{Cd}$ equals the product of the amount of earthworms consumed and the Cd concentration in these worms (equation 6.1).

Figure 6.2a presents the time it takes to reach the state of kidney lesion for badgers feeding on earthworms that live in soils with Cd concentrations in the range of 0.1 to 5.7 mg kg$^{-1}$ and pH values of 3.5, 6, and 7.2, respectively. Figure 6.2a indicates that badgers run the risk to suffer from kidney lesion within two to six years when they forage in areas with acid soils (dotted line in Fig. 6.2a). On the other hand badgers living in areas with a high pH are not expected to run a risk on kidney lesion (solid line in Fig. 6.2a). Figure 6.2b shows the time it takes badgers to reach kidney lesion within their actual distribution in The Netherlands. The grid cells in Figure 6.2b represent 5x5 km blocks. The time it takes to reach kidney lesion in these grid cells is based on the mean Cd and pH values of the 1x1 km blocks underlying the grid cell where one or more inhabited badger setts are situated. Although badgers can reach maximal ages of 10 to 15 years (Kruuk & Parish 1987; Cheeseman et al. 1987; Graf & Wandeler 1989) the largest part of the Dutch badger population does not seem to reach ages above six years (Müskens & Broekhuizen unpublished results). But even if badgers

![Graph](image-url)
do not reach ages above six years they still run a risk to suffer from kidney lesion in many parts of their range (Fig. 6.2b). Especially near the Kempen (known for its metal smelters) and in the Veluwe areas badgers run a high risk to suffer from kidney lesion. The results (Figs. 6.2a, b) are supported by Cd levels detected in kidneys of badgers killed by traffic on the Veluwe and near the river Meuse (Ma & Broekhuizen 1989). A high percentage of these road victims, all less than six years old, had reached the critical Cd level of 200 µg Cd g⁻¹ (Ma & Broekhuizen 1989). Moreover, badgers found near the Meuse riverbanks had reached the critical kidney level at the age of two to five years as indicated by Figure 6.2b. In contrast, the results of Figure 6.2b in the Veluwe area are not supported by Ma & Broekhuizen 1989. Figure 6.2b indicates that in the Veluwe the critical level for Cd by badgers is reached within two to four years, but analyses of kidney Cd contents of traffic victims show that in this area the critical Cd level is not reached (Ma & Broekhuizen 1989). The risk to suffer from kidney lesion in the Veluwe area is seemingly lower than assessed by the bioaccumulation calculation (equation 6.2). This difference in calculated and actual risk possibly results from the fact that in poor sandy soils, like the Veluwe, earthworms are scarce and so the diet composition of badgers in these areas consists of less than 30% earthworms. This argument is confirmed by analyses of the stomach contents of badgers killed by traffic in the Veluwe area (Müskens & Broekhuizen unpublished results).

**Food shortage as an indirect effect of Cu on the badger**

Badgers that forage on soils with a high Cu concentration may be confronted with a shortage of earthworms. Many earthworm species are sensitive to Cu leading to low densities in soils polluted with Cu (Edwards & Lofty 1975). In orchards where Cu is frequently applied densities of the earthworm species *L. rubellus* and *L. terrestris* were found to be very low (Van Rhee 1967). These low densities result from effects of Cu on the individual earthworms. As indicated by laboratory studies (Ma 1983b, 1988) Cu reduces the individual growth and cocoon production in earthworms. The impact of Cu on the population level can be assessed with population dynamic models that track the growth, reproduction and mortality of all the individuals in the population and integrate the individual effects into effects on the population growth rate (Klok & De Roos 1996; Klok et al. 1997) or the expected population density (Bavec o & De Roos 1996).

To assess the impact of Cu on food abundance for badgers, that is the population density of earthworms, a physiological structured population model (De Roos 1997) was used. Maturation in earthworms depends more on size than on age (Cluzeau & Fayolle 1989; Ma 1984, 1988); therefore the size of the individuals has been taken as the main structuring variable in the model. The Dynamic Energy Budget (DEB) formulation (Kooijman & Metz 1984) was used to describe the growth in size and
Figure 6.2b

Expected time in years until the critical level of kidney lesion (200 $\mu$g Cd g$^{-1}$ dry weight) is reached in badgers, for the distribution of 1995.
reproduction of the individual earthworms. The DEB model relates food intake, maintenance, growth, and reproduction of the individuals in a straightforward manner. Food intake is assumed to be proportional to surface area \( I^2 \) whereas maintenance and growth are proportional to body mass \( I^3 \). Here and below, \( I \) is interpreted as the cubic root of individual weight, which is assumed to be proportional to length (Kooijman & Metz 1984). The ingested energy is distributed in a fixed proportion \( K \) to maintenance and growth and (1-\( K \)) to reproduction. Hence reproduction is proportional to food intake, which scales with surface area \( I^2 \). Maintenance always takes precedence over growth. This leads, under abundant food conditions, to an attenuating growth curve. Growth stops when individuals reach the size where the ingested energy is just enough to fulfil the maintenance needs. Reproduction starts if the individuals have attained a certain size \( l_{ad} \). Furthermore, individuals of different sizes are assumed to have the same allometric relations.

The size of the individuals at age \( a \) is given by:

\[
l(a) = l_m - (l_m - l_b)e^{-\gamma a}
\]  

(6.3)

where \( l_m \) equals the maximal attainable size, \( l_b \) the size at birth, and \( \gamma \) the individual growth rate.

The reproductive rate at the age \( a \) is a function of the individual size, and is given by:

\[
m(a) = r_m \cdot l(a)^2 \quad \text{if} \quad l(a) \geq l_{ad}
\]

(6.4)

where \( r_m \) equals the maximum attainable reproductive rate per unit of surface area.

Experimental data show that under Cu treatment both growth and reproduction of individual earthworms decrease (Ma 1983a, 1983b). Depending on how the toxic agent influences the energy budget of the individual (decrease in assimilation due to e.g. impairment of digestion, increase in maintenance resulting from detoxification etc. see Kooijman & Metz 1984) the parameters \( l_m, \gamma, \) and \( r_m \) in equations 6.3 and 6.4 will change. It is assumed that the effects of Cu on the individual growth and reproduction assessed in bioassays by Ma (1983a, b) mainly result from increased maintenance costs. This leads to an increase in \( \gamma \) and a decrease in \( l_m \) since the individual growth rate \( \gamma \) is proportional to the maintenance requirements per weight unit, whereas the maximal attainable size \( l_m \) is inversely proportional to this factor. Equation 6.4 indicates that with a decrease in individual growth (equation 6.3) the reproductive rate will decrease as well. However, the reductions in reproductive output given by Ma (1983b) are larger than can be explained on the basis of this assumption about increased maintenance requirements. Therefore, it was furthermore assumed that Cu increases the energy requirement to produce a single cocoon which leads to a reduction in \( r_m \), the maximum
attainable reproductive rate (see Appendix 6.1 for the estimation procedure of $l_m$ and $\gamma$, and $r_m$).

The survival of earthworms living under optimal conditions only depends on age and can be approximated by a constant mortality rate $\mu_0$ equal to 0.001 day$^{-1}$ (Lakhani & Satchell 1970). Under field conditions earthworms have an increased mortality resulting from predation. In this paper it was assumed that earthworm populations are regulated by predation to a density of 100 individuals m$^{-2}$ when Cu is at the background level. The survival of earthworms can be described by:

$$S(a) = e^{-(\mu_0 + \mu_1 \cdot N)a}$$

(6.5)

were $\mu_1$ equals the predation pressure, and $N$ the earthworm population density.

Given an equilibrium earthworm density of 100 individuals m$^{-2}$ when Cu is at the background level the predation pressure $\mu_1$ can be calculated using the following equations, which represent the equilibrium conditions for the structured population model (De Roos 1997):

$$\int_{a}^{b} m(a) \cdot S(a) \, da = \int_{a}^{b} r_m \cdot l(a)^2 \cdot e^{-(\mu_0 + \mu_1 \cdot N)a} = 1$$

(6.6)

where $\alpha$ equals the duration of the juvenile period (the time it takes the individuals to grow from $l_b$ to $l_{ad}$), $\beta$ the maximal age, and $\bar{N}$ the equilibrium earthworm density.

The equilibrium density attained by earthworms in Cu polluted soils will decrease with the Cu level, whereas Cu stress reduces individual growth and reproduction in earthworms. These reductions, which are assumed to result from increase in maintenance costs and energy requirement to produce a single cocoon (see above), lead to increase in the juvenile period ($\alpha$) and the parameter $\gamma$, and to decrease in the parameters $l_m$ and $r_m$. With predation pressure $\mu_1$, corresponding to the equilibrium population density of 100 individuals m$^{-2}$ when Cu is at a background level, the equilibrium density under Cu stress can be assessed using equation 6.6. In the calculations the environmental conditions (food, temperature etc.), apart from Cu stress, are assumed to be constant and equivalent to those given by Ma (1983a, b). A more detailed description of a similar model can be found in Bavec & De Roos 1996.

The population dynamical model was used to calculate the population density and biomass of earthworms under Cu levels in the range of Dutch Cu soil concentrations, given sublethal effects of Cu based on laboratory bioassays in sandy soil (Table 6.3). Not all sizes of earthworms are available to badgers. Analyses of stomach contents indicate that badgers only consume earthworms with sizes larger than 5 cm corresponding to a weight of around 600 mg (Müskens & Broekhuizen unpublished
data). In the model it therefore is assumed that badgers only prey upon earthworms that have a size of \( l = 8.5 \text{ mg}^{1/3} \) or more.

Figure 6.3a presents the biomass production of earthworms available to badgers (based on earthworms with sizes larger than 8.5 mg \(^{1/3}\)) as a function of Cu soil contents. At the background Cu soil concentration (13 mg kg\(^{-1}\)) the earthworm population exists of 100 individuals per square meter. At this Cu level the biomass production of individuals sized larger than 8.5 mg \(^{1/3}\) equals 480 mg per square meter (see Fig. 6.3a). The figure furthermore indicates that the biomass production of earthworms is reduced by 50% if the Cu concentration is increased to a level of 100 mg kg\(^{-1}\). Figure 6.3b shows how these results translate into decreases in biomass production of earthworms available to badgers within their actual distribution. Each grid cell indicates an area of 5x5 km. The decrease in these grid cells is equal to the highest decrease in one of the 25 underlying 1x1 km cells. Only those 1x1 km cells are included with at least one inhabited badger sett situated in the cell. Especially at the fringes of their range, the Gooi area in the west, the Reestdal area in the north and the Kempen area in the south, badgers are expected to suffer from food shortage resulting from decreased earthworm densities. Unfortunately, these results cannot be confronted
Figure 6.3b
Reduction in earthworm biomass production, induced by Cu, in areas occupied by badgers in 1995.
with field data because population densities of earthworms in the three areas mentioned, and most regions in The Netherlands, are unknown. But if the results of Van Rhee (1967) and Ma (1988) which both show that Cu does have a negative effect on the density of earthworms are considered, it can be expected that badgers suffer from decreased food availability in some parts of their range.

Discussion

Effects on individuals

The results indicate that, given the soil concentrations of Cd and Cu in The Netherlands, effects of these heavy metals on the badger can be substantial. In some parts of their range badgers run the risk to achieve kidney lesion within two to four years (direct effect Cd; Fig. 6.2b), and food shortage (indirect effect of Cu; Fig. 6.3b).

The bioaccumulation calculation predicts kidney lesion as a result of Cd intoxication. These results are conservative in the way that they only refer to Cd uptake by badgers resulting from consumption of earthworms. The addition of Cd reaching the badger by other means will only increase the risk to achieve kidney lesion.

Kidney lesion often has been put forward as one of the first serious effects of Cd intoxication in mammals and birds (Schuhammer 1987). However, to what extent kidney lesion reduces the vital rates (growth, reproduction, and survival) in badgers is unclear. It may be expected that necrosis of kidney tissue, which leads to a decrease in the filter capacity of the kidney (Nicholson et al. 1983; Goyer et al. 1984; Schuhammer 1987), also leads, next to passive leaking of Cd, to loss of nutritional and essential elements. This loss involves a decline in growth, reproduction, and survival of the individual. If the individual compensates for the loss by increase of food intake, the animal will spend more time in search of food, and roam larger areas. In a country like The Netherlands with its dense infrastructure this can lead to an increase in traffic mortality.

As with kidney lesion also the impact of food scarcity on the vital rates is difficult to quantify. When earthworm density is low, badgers may change their diet composition, or spend more time in searching for earthworms, and thus roam larger areas. Changing the diet composition will lead to a decrease in body weight (Kruuk & Parish 1983), whereas increasing the time spent and the area searched for earthworms enhances the risk of traffic mortality. Therefore, both these alternatives have a negative effect on the vital rates of the badger.
Effects on the population

How effects of Cd and Cu will reveal at the population level depends on the impact of these effects on the growth, reproduction, and survival of the individuals that make up the population. Decrease in these vital rates reduces the growth rate and the survival probability of a population. Whether decrease in the vital rates will also become manifest in a measurable population parameter like density depends on the dynamic interaction of the species with its (biotic and abiotic) environment. Model studies on the effect of toxic substances on the shrew Sorex araneus demonstrated that a decrease in the vital rates had a negligible effect on the expected population density whereas the population survival time decreased drastically (Klok & De Roos 1998). In Sorex the high mortality caused by the contest competition for territories in autumn conceals any effect of other stress factors on the vital rates as long as the lifetime reproductive success of the individuals is larger than 1 (individuals can replace themselves). So the effects of toxicants will not show up in the population density until individuals can no longer replace themselves, leading to a sudden, drastic decline in the population density. Also in the badger, effects of toxicants may not become apparent in the population density. Badgers live in hierarchical social groups, where only one female puts effort in reproduction (Kruuk & Parish 1983; Broekhuizen et al. 1994). If this female dies, one of the other adult females of the social group will take her position. Therefore, the reproductive output of a badger group will remain more or less constant over time.

Multiple stress

Acidification of the soil in combination with Cd pollution is an example of multiple stress for the badger. On the one hand a decrease in pH will lead to lower densities of earthworms (Edwards & Lofty 1975) and thus food shortage for badgers. On the other hand with low pH levels Cd is more mobile in the soil and the uptake of Cd by earthworms is increased (Ma et al. 2000) with a greater risk of poisoning for badgers. Figure 6.2a indicates the importance of soil pH for the risk of kidney lesion in badgers. In 95% of the Dutch soils the Cd concentration is below 1.0 mg kg⁻¹ (Tiktak et al. 1999a, b). With a Cd soil concentration of 1.0 mg kg⁻¹ badgers will reach the critical level of kidney lesion before the age of six (Fig. 6.2a). For soils with Cd concentrations below 1.0 mg kg⁻¹ badgers are not expected to suffer from kidney lesion within the life span of six years. But with acidification of the soil kidney lesion may even be expected in areas with Cd soil concentrations below 1.0 mg kg⁻¹.

Also the combination of Cu enrichment of the soil and habitat fragmentation is an example of multiple stress for the badger. When badgers counter the indirect effect of Cu (food shortage) by putting more effort into foraging, they increase the probability of getting killed by traffic. Traffic mortality in The Netherlands is high,
yearly around 20% of the badger population falls victim to traffic (Broekhuizen & Derckx 1996). This high mortality has been attributed to the fragmentation of the badger habitat. Whether this high traffic mortality is caused by fragmentation alone or in combination with other factors deteriorating the habitat quality, such as poisoning and acidification, cannot be deduced from the mortality itself.

Although it is difficult to indicate to what extent the quality of the habitat plays a role in the traffic mortality, it seems reasonable to assume that increase in the quality of the habitat will decrease the risk of traffic mortality for the individuals. In The Netherlands with its high density of infrastructure, next to countering fragmentation by connecting habitat patches, also increasing the quality of these patches seems to be a good option to enhance the survival probability of badger populations.

Conclusions

Apart from restrictions on hunting and poisoning, conservation actions in The Netherlands intended to increase the survival probability of badger populations have been directed mainly towards decreasing the traffic mortality. Up to 1996 more than 200 badger tunnels were constructed under roads (Broekhuizen & Derckx 1996). The high traffic mortality has been attributed to the fragmentation of badger habitat. To what extent habitat quality promotes traffic mortality is unknown.

This paper shows that within the distribution area of the badger, Cd and Cu soil concentrations are expected to give rise to negative effects. Figure 6.2b indicates that badgers in a large part of their range are threatened to suffer from kidney lesion, whereas badgers may experience food shortage in some parts of their distribution area (see Fig. 6.3b). To what extent these effects reveal themselves at the population level cannot be easily inferred from field data. Ecological interactions at the population level may conceal effects of decreased habitat quality.

References

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Appendix 6.1

Laboratory studies (Ma 1983b, 1988) indicate that Cu reduces the individual growth and cocoon production in the earthworm species *Lumbricus rubellus*. In these studies earthworms were cultured in soils polluted with four Cu levels; 13, 60, 145, and 362 mg Cu kg\(^{-1}\) soil. It was assumed that these reductions in growth and cocoon production result from increase in maintenance costs. This assumption leads to an increase in \(\gamma\) and a decrease in \(l_m\) since the individual growth rate \(\gamma\) is proportional to the maintenance requirements per weight unit, whereas the maximal size \(l_m\) is inversely proportional to this factor. Under Cu stress the maximal size can now be described by:

\[
l_{m,Cu} = \frac{l_m}{(1 + (Cu - NOEC)\rho_M)} \tag{6.A1}\]

the individual growth rate by:

\[
\gamma_{Cu} = \gamma(1 + (Cu - NOEC)\rho_M) \tag{6.A2}\]

and the individual growth by:

\[
l(a)_{Cu} = l_{m,Cu} - (l_{m,Cu} - l_0)e^{-\gamma_{Cu}a} \tag{6.A3}\]

where Cu equals the external copper concentration, \(NOEC\) the copper concentration where growth is not reduced, and \(\rho_M\) the effect parameter on the maintenance requirements.

To estimate the values of \(\gamma\), \(l_m\), and \(\rho_M\), growth curves describing the growth of individuals under copper stress of 13, 60, 145, and 362 mg Cu kg\(^{-1}\) soil respectively (equation 6.A3) were fitted simultaneously to the four experimental data series on the increase in individual weight over time of earthworms living under Cu levels of 13, 60, 145, or 362 mg Cu kg\(^{-1}\) soil. The \(NOEC\), the copper concentration where growth is not reduced, is fixed at the background copper concentration of 13 mg kg\(^{-1}\) soil.

Estimated values for \(\gamma\), \(l_m\), \(\rho_M\), and \(NOEC\) are 0.0071 day\(^{-1}\), 16.66 mg\(^{1/3}\), and 0.0029 respectively.

Equations 6.3 and 6.4 indicate that with a decrease in \(l_m\) and an increase in \(\gamma\) also the reproductive output decreases. However, the decrease in \(m(a)\) given by Ma (1983b, 1988) is more drastic than can be explained by the assumption of increase in maintenance costs. Therefore, it was assumed that next to an increase in maintenance costs also the energy investment to produce a single cocoon increases with Cu stress. This leads to a reduction in \(r_m\) (see equation 6.4). The experimental data on the
number of cocoons produced per day per unit of surface area under different Cu treatments were fitted by the curve:

\[ r_{m,Cu} = \frac{r_{m,0}}{1 + RhoR \cdot Cu} \]

where \( r_{m,0} \) equals the maximal reproductive rate per unit of surface area, and \( RhoR \) the effect parameter on energy investment to produce a single cocoon.

Estimated values for the parameters \( r_{m,0} \), and \( RhoR \) are 0.00126 d\(^{-1}\)mg\(^{-2/3}\) and 0.0189, respectively.