MAMMALS AS BIOLOGICAL MONITORS OF ENVIRONMENTAL METAL LEVELS

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Abstract. Wild mammals can be valuable biological monitors of environmental gradients of metal concentrations. The choice of a particular species for a biological monitor must be based upon the circumstances of each study including species availability, the metals to be examined, area, and the study objectives and priorities. Ideally, a biological monitoring study should be designed to obtain and make use of the optimum amount of available information by complementing existing environmental studies, or through the simultaneous collection of other environmental data.

1. Introduction

A number of studies have reported metal concentrations in wild mammals. A large proportion of these studies generally compare metal levels in animals from more than one location, often where one or more sites is known, or suspected, to be contaminated. In such instances the species in question is being utilized, either knowingly or otherwise, as a biological monitor of environmental metal loading. There have been some excellent reviews and texts written on the general subject of biological monitoring (Phillips 1980; Jenkins 1980; Martin and Coughtrey 1982). However, as Martin and Coughtrey (1982) point out, in comparison with aquatic organisms, the potential of terrestrial organisms as monitors has not been discussed adequately. Information on the use of terrestrial mammals as biomonitors is even more limited. This paper provides an overview of approximately 50 studies reporting metals data in a variety of wild mammals species. Some of the factors influencing metal uptake in wild mammals are discussed, and the use of mammals as biomonitors of environmental metal loading and availability is evaluated.

2. Biomonitoring Concepts

Effective environmental management requires knowledge of the transport and fate of contaminants in natural exosystems. Connell and Miller (1984) state that the objectives of environmental monitoring can be met by focusing on two aspects: (1) Monitoring the pollutant in different parts of the environment (factor monitoring) and, (2) Monitoring the effects of the pollutant on the natural ecosystem and associated biota (target monitoring). Factor monitoring generally involves chemical and physical measurements, where as target monitoring is concerned with the response of biological systems. Biologists and toxicologists are, quite naturally, principally interested in the latter aspect of environmental monitoring.

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There are many reasons for conducting a biological monitoring study. Munn (1973) provides insight into the general objectives of environmental monitoring:

(a) increase quantitative knowledge of natural and man-made changes in the environment;

(b) increase understanding of dynamic balance in ecosystems;

(c) provide early warning of significant environmental changes;

(d) check the effectiveness of established regulatory mechanisms.

To this list I would add two more values of biomonitoring of metals taken from Jenkins (1980):

(e) define critical pathways of pollutants to humans from water, air and food;

(f) integrate biological exposure of toxic trace elements with physical and chemical measurements in the environment.

It should be recognized that many of the initial objectives of biomonitoring and existing monitoring programs were designed from the viewpoint of identifying potential hazards to humans. However, researchers are becoming increasingly concerned with the impact of toxic substances to natural ecosystems and populations of wild organisms.

Grodzinski and Yorks (1981) establish that there are three basic categories for the use of plants and animals as bioindicators of environmental quality:

(1) a scale indicator species (presence – absence);

(2) true indicators (exhibiting damage proportional to dose);

(3) accumulators (of potentially toxic substances).

The concepts of biological monitoring are further simplified by Jenkins (1980) who states there are only two fundamental biomonitoring methods concerning metals in the environment. The first method is to measure the accumulation or concentration of toxic metals in selected monitoring organisms. The second method is the measurement of actual impacts or effects of toxic trace elements on individual, or populations of, organisms. The second technique can involve a variety of measurements including diversity indices, physiological and biochemical indices, calculation of reproductive rates, population age structure, etc., suited to a specific need. Accurate quantification of such parameters in wild mammals, however, is often hampered by sampling logistics due to limited availibility of specimens and lack of reliable information concerning the normal state of the wild species in question. Furthermore, measurements of biological impacts will be confounded by the effects of other pollutants present in the environment as well as natural stresses (ie. disease, climate, starvation) and natural population fluctuations. In contrast, the direct measurement of a metal or other contaminant within the organism can provide accurate data on the environmental availability, mobility and fate of the element within a specific ecosystem. It must be emphasized, however, that while the measurement of metal concentrations in tissues of wild organisms provide valuable environmental data, little biological or ecological significance can be attached to the levels without comparative toxicological or experimental observations.

2.1. Some Case Studies of Mammals as Biological Monitors

A search of the primary literature revealed approximately 50 studies which utilized wild mammals as biological monitors of environmental metal loading (Table I). There are a number of published reports of metal levels in a variety of mammal species but from only one location. These studies do not measure animal body burdens along a gradient of metal exposure and are, therefore, not true biomonitoring investigations. The following section contains a brief review of studies which encompass a range of mammal species, geographic conditions and pollution sources.

Except in instances of obvious environmental contamination from a point source, it is seldom possible to attribute adverse biological effects to a single pollutant in the field. For instance, the combustion of fossil fuels and certain mining and smelting activities release metals as well as sulphur and nitrogen gases and particulate matter into the environment. Kucera (1980, 1981) investigated the effects of smelter emissions on wild rodents near Flin Flon and Thompson, Manitoba (Canada). Tissue levels of As, Pb, Cu, Zn, and to a lesser degree Co and Cd in deer mice (*Peromyscus maniculatus*) decreased with distance away from the Flin Flon smelter and approached background levels 25 km away (Kucera, 1980). Near the Thompson smelter, tissue levels of As and Ni in deer mice were elevated within a 30–50 km distance from the emission source (Kucera, 1981). Examination of blood parameters revealed that hemoglobin (Hb) and packed cell volume (PCV) levels were elevated near the Flin Flon smelter, while Hb concentrations were higher in deer mice collected near the Thompson smelter.

Kucera (*ibid*) also observed that the catch per-unit-effort of red-backed voles (*Clethrionomys gapperi*) was markedly lower near the smelters. While the number of animals caught was too limited for statistical analysis, a similar trend was reported at both the Flin Flon and Thompson smelters and repeated in subsequent years. No similar pattern was noted for deer mice, indicating that red-backed voles may be more susceptible to the emissions than other species. The author suggests that if these trends continue, smelter emissions could alter the abundance and composition of small mammals, with possible lowering of productivity of higher animals feeding on them.

Small mammals have been utilized to examine the environmental loading of metals from a variety of activities such as chlor-alkali plants (Bull *et al.*, 1978; Haschek *et al.*, 1979), agriculture (Fimreite *et al.*, 1970), sewage-sludge treatment to fields (Beardsley *et al.*, 1978; Anderson *et al.*, 1982; Anthony and Kozlowski, 1982), and mine tailing disposal sites (Roberts and Johnson, 1978; Hunter and Johnson 1982; Johnson *et al.* 1978; Roberts *et al.*, 1978). Small mammals were generally considered useful biomonitors for specific metals (Table I), but, Beardsley *et al.* (1978) concluded that field voles were not a suitable monitoring species at a sewage-farm site since the response of metal levels in tissues of the voles was too small and too erratic. Martin and Coughtrey (1982), however, felt this conclusion was too general, and that closer examination of the data of Beardsley *et al.* (1978) indicate the differences in liver Cd and Cu concentrations do reflect the differences in metal content of the various habitats.

Several studies used small mammals to investigate the dispersal of metals from

TABLE I

Mammals used as biological monitors of environmental gradients of metal concentrations

Species	Pollution Source	Tissue[a]	Metal	Value as bio-monitor[b]	Reference
mink, otter	polluted vs non-polluted areas	L, K, Br	Hg	E	Kucera, 1983
mink	polluted vs non-polluted	H, L, K, M, Br.	Hg	E	Sheffy, 1977
otter	polluted vs non-polluted	H, L, K, M, Br	Hg	E	
raccoon	polluted vs non-polluted	H, L, K, M, Br	Hg	E	
muskrat	polluted vs non-polluted	H, L, K, M, Br	Hg	P	
red fox	polluted vs non-polluted	H, L, K, M, Br	Hg	P	
beaver	polluted vs non-polluted	H, L, K, M, Br	Hg	P	
Polar bear	geological differences	H	Hg	E	Eaton and Farant, 1982
bobcats	polluted vs non-polluted	H	Hg	G	Cumbie and Jenkins, 1975
raccoons	polluted vs non-polluted	H	Hg	E	
gray fox	polluted vs non-polluted	H	Hg	P	
white-tailed deer	polluted vs non-polluted	H	Hg	P	
opossum	polluted vs non-polluted	H	Hg	P	
bobcat	polluted vs non-polluted	H, L, M	Hg	G	Cumbie, 1975b
raccoon	polluted vs non-polluted	H, L, M	Hg	E	
coyote	mineralized vs non-mineralized area	H	Hg	P	Huckabee *et al.*, 1973
small rodents	mineralized vs non-mineralized area	H	Hg	G	
muskrats	zinc mine, agricultural and industrial pollution	L, K	Zn	E	Everett and Anthony, 1977
		L, K	Cd	E	
		L, K, Bo	Pb	P	
		L, K	Cu	G	

Table I (Continued)

Species	Pollution Source	Tissue[a]	Metal	Value as bio-monitor[b]	Reference
muskrat	smelter	L, K, M, D	M*	P	Radvanyi and Shaw, 1980
	*Multielements – Al, Pb, Cd, Fe, Mg, Zn, As, Se, Cu, Ni				
muskrats	industrial and municipal waste	L, K	Pb	E	Erickson and Lindzey, 1980
		L, K	Cd	P	
raccoons	urban	L	Pb	E	Diters and Nielson, 1978
raccoons	regional differences	H, L, M	Hg	E	Bigler *et al.*, 1975
raccoons	none	K	Pb, Zn, Cd	G	Hoff *et al.*, 1977
moose	regional differences	L, K	Cd	E	Mattson *et al.*, 1981
moose	regional differences	L, K	Cd	G	Frank *et al.*, 1981
roe deer, hare	regional differences	L, K	Pb	G	
moose	regional differences	H	M*	G	Franzmann *et al.*, 1977
	*Multielements – Cd, Pb, Cu, Fe, Mn, Zn, Ca, K, Mg, Na				
roe deer	steel works	A	Cr, Pb, Cd	E	Sawicka-Kapusta, 1979
		A	Fe, Zn	G	
		A	Mn, Cu, Ni	P	
		A	S		
white-tailed deer	different ecoregions and counties	K	Pb, As, Hg, Cd	G	Kocan *et al.*, 1980
Cottontail rabbits	regional differences	K, M	Pb, Cd, Hg, As	E	Lynch, 1973
gray squirrels	urban vs rural	K	Zn, Pb, Cd	G	McKinnon *et al.*, 1976
field vole	sewage sludge	L, K, Br	Zn, Cu, Mn Cd, Pb, Cr	P	Beardsley *et al.*, 1978
		L	Cd, Cu	G	
		K, Br	Cd, Cu	P	
field vole	mine-tailings	WB	Pb, Cd	E	Roberts and Johnson, 1978
		WB	Zn	P	
shrew	mine-tailings	WB	Pb, Cd	E	
		WB	Zn	P	
small mammals	mine-tailings	K	Pb	E	Johnson *et al.*, 1978
		L, Bo	Pb	G	
		L, K, M, H	Zn	P	
		L, K	Cd	E	
		M, Br, Bo	Cd	P	

Table I (Continued)

Species	Pollution Source	Tissue[a]	Metal	Value as bio-monitor[b]	Reference
meadow voles	sewage-sludge	L, K	Cd	G	Anderson *et al.*, 1982
		L, K	Pb, Zn, Cu	P	
small mammals	old orchards	L, K, Bo	Pb	E	Elfving *et al.*, 1978
meadow vole	sewage-sludge	L, K	Ni, Cu, Zn, Cr, Pb, Co	P	Anthony and Kozlowski, 1982
white-footed mice		L, K	Pb, Cd	G	
		L, K	Cu, Zn, Ni, Cr	P	
deer mice	Zn-Cu mine	WB	Zn, Cu, Cd, Pb	E	Smith and Rongstad, 1982
		WB	Ni, Hg, As		
meadow voles	Zn-Cu mine	WB	Zn, Cu, Cd, Pb	P	
		WB	Ni, Hg, As	P	
deer mice	smelter	C	Pb	E	Kucera, 1980
		V	Cd	G	
		C	Co	G	
		Hi	As, Cu, Zn	E	
		Hi	Ni	P	
		V	Fe	P	
deer mice	smelter	Hi	As	E	Kucera, 1981
		Hi	Ni, Cu	G	
		Hi	Zn	P	
		V	Mn, Fe	P	
		C	Co, Pb, Cd	P	
woodmice	chlor-alkali plant	H, K, L, Br			
		M	Hg	G	Bull *et al.*, 1977
			Hg	P	
bank voles	chlor-alkali plant	L, K, Br, H, M	Hg	G	
small mammals	highway traffic	WB	Pb	E	Goldsmith and Scanlon, 1977
deer mice	highway traffic	L, K, Bo	Pb	E	Welch and Dick, 1975
		Br	Pb	P	
shrews, voles	revegetated mine site	WB, L, K,	Cd	E	Andrews *et al.*, 1984
		T, Bo	Cd	P	
gray squirrels	urban vs. rural	H	Hg	G	Jenkins *et al.*, 1980
ground squirrels	Hg-treated fields	L	Hg	G	Fimreite *et al.*, 1970
small mammals	geographic variation	L	Cd	E	Sharma and Shupe, 1977
		Bo	Cd	P	
		L, Bo	Pb	P	
		L	As	P	
white-tailed deer	uranium mines	L	Mb, Cu	P	King *et al.*, 1984
white-tailed deer	industrial complex	Bo,T	F	E	Karstad, 1967

Table I (Continued)

Species	Pollution Source	Tissue[a]	Metal	Value as bio-monitor[b]	Reference
white-tailed deer	geographic variation	L	Cu, Mg, Mn, Pb As, Hg, Ni, Zn	P–G	Woolf *et al.*, 1982
roe deer	Hg-treated fields	M, L, K	Hg	G	Krynski *et al.*, 1982
roe deer	Al plant and generating station	H	As, Cd, Pb, Zn, F	E	Mankovska, 1980a
roe deer	Al plant	T	F	E	Mankovska, 1980b
		T	As, Pb, Cd	G	
cows	Pb smelter	Bl	Pb	E	Karicic *et al.*, 1984
cows	geographic variation	L, K	As, Cd, Se	G	Kramer *et al.*, 1984
		L, K	Cu, Zn, Cr	P	
		M	all metals	P	
cows	atmospheric deposition	K	Cu, Cd, Pb	E	Gydesen *et al.*, 1981
		K	Zn	P	
small mammals	highway traffic	WB, L	Pb	G	Jeffries and French, 1972
small mammals	highway traffic	WB	Pb	G	Getz *et al.*, 1977
deer mice	highway traffic	Bo	Pb	E	Mierau and Favara, 1975
		L, K, B	Pb	G	
small mammals	highway traffic	WB	Pb	G	Quarles *et al.*, 1974
small mammals	highway traffic	L, K	Pb	G	Williamson and Evans, 1972
small mammals	urban vs. rural	H	Pb	G	Raymond, 1975
small mammals	old orchards	Bo	Pb	E	Haschek *et al.*, 1979
		L, K	Pb	G	

[a] L = Liver; K = Kidney; M = muscle; Br = brain; Bo = bone; H = hair; Bl = blood; Hi = hide; C = mcarcass; C = viscera; D = duodenum; WB = whole body; A = antlers; T = teeth

[b] E = excellent; G = good; P = poor

Value as biomonitor is a rating based on data evaluation by this author and discussion in original reference.

abondoned mine and industrial sites in Britain (Roberts and Johnson 1978; Johnson *et al.*, 1978; Roberts *et al.*, 1978). Metal ratios in native fauna were sometimes inconsistant with those of local soil and vegetation. For example, despite at 17 fold increase in vegetation Zn levels from the control to the mine site, body Zn levels in voles (*Microtus agrestus*) only increased by 50%. In contrast, levels of Pb were up to 15 times greater in animals from the mine site compared with the control site (Roberts and Johnson 1978). The low transfer potential of Zn may be associated with its essential role and effective homeostatic regulation in biological systems. Hunter and Johnson (1982) also reported a much greater food chain transfer potential for Cd than for Cu in small mammals despite relatively higher environmental loading of Cu. These authors suggest that measurement of metal levels only in soils and vegetation was not an adequate indicator of metal availability owing to contrasting mobility and accumulation potential.

Numerous studies have investigated the effect of automobile exhaust on Pb levels in small mammals living near highways (Lutmer *et al.*, 1967; Mierau and Favaria, 1975; Welch and Dick, 1975; Goldsmith and Scanlon, 1977; Williams and Evans, 1972). The general findings of these studies are that Pb levels are elevated in soils, vegetation and small mammals living within about 30 m of major highways. There are no indications that even very elevated Pb levels are toxic to individuals or populations of small mammals, but rapid recruitment from nearby areas could mask any adverse biological effects (Quarles *et al.*, 1977). Differences in species Pb levels have been noted and are generally attributed to dietary differences.

Jefferies and French (1972) collected vegetation and small mammals from roads in England with varying traffic densities. Vegetation adjacent to a major highway showed the highest degree of Pb contamination (226 $\mu g\, g^{-1}$ wet wt) compared with vegetation growing in unpolluted fields (10 – 25 $\mu g\, g^{-1}$). Mean Pb levels in small mammals decreased from 2.26 $\mu g\, g^{-1}$ on the verge of a major highway, to 1.92 $\mu g\, g^{-1}$ on the verge of minor roads, to 1.32 $\mu g\, g^{-1}$ at an uncontaminated woodland site. Field voles contained significantly higher total-body levels of Pb (3.14 $\mu g\, g^{-1}$) than either bank voles or woodmice, which contained 1.89 $\mu g\, g^{-1}$ and 1.62 $\mu g\, g^{-1}$, respectively, on roadside verges. In addition, body concentrations of Pb were higher in females than males of each species. The authors suggest that species differences in Pb levels are probably due to diversity of eating habits. Field voles feed primarily on grass, the most heavily contaminated material, compared with the diet of seeds, buds, nuts and animal matter of the other species. Other studies, however, reported that carnivorous small mammals contain higher Pb levels than herbivorous species (Quarles *et al.*, 1977).

Scanlon (1979) compared Pb levels in soils, plants, insects, and small rodents collected adjacent to major highways of different traffic volume with levels from remote forested areas in the United States. Significant differences in Pb concentrations between locations were found in least shrews (*Crytotis parva*), meadow voles (*Microtus pennsylvanicus*) and white-footed mice (*Peromyscus leucopus*). In all cases, animal Pb levels were related to traffic volume. Vegetation Pb levels differed significantly both between sites, and with season of collection. Within areas, insectivorous shrew species tended to have higher Pb concentrations than herbivourous species. For example, Pb concentrations in short-tailed shrews (*Blarina brevicauda*) were 87.3 $\mu g\, g^{-1}$ (dry wt) from an area having a traffic volume of 100 000 vehicles per day, and only 1.7 $\mu g\, g^{-1}$ in animals from the control area. In comparison, Pb concentrations in white-footed mice ranged from 16.3 $\mu g\, g^{-1}$ in the high traffic area to 0.9 $\mu g\, g^{-1}$ in the control area.

Sharma and Shupe (1977) determined that mammalian liver was an excellent indicator of environmental Cd levels. Rock squirrels (*Spermophilus varigatus*) and pack rats (*Neotoma cineren*) were collected from 18 different locations representing a wide range of metal concentrations in soil. Cadmium concentrations in liver tissue were highly correlated with both vegetation and soil Cd levels, while bone Cd levels were not related to environmental concentrations. Similarly, neither Pb or As levels in animal tissue were related to environmental concentrations.

Andrews *et al.* (1984) used short-tailed field voles (*Microtus agrestis*) and common shrews (*Sorex graneus*) as biological monitors of Cd availability between a control area and a revegetated mine site. Total body burdens of Cd were significantly higher in both species from the mine site compared with the control site. Also, at the mine site, Cd levels in the insectivorous shrew were 28 times higher than in herbivorous voles, reflecting differences in dietary levels and availability of Cd to the consumer. The authors report that, in general, bony tissues were not as suitable as soft tissues, especially liver and kidney, in reflecting differences in environmental Cd burdens between sites.

Everett and Anthony (1977) found that levels of Cd in muskrat liver and kidney were related to Cd levels in plants at 4 different sites, and suggest that the muskrat is a valid indicator of Cd pollution in aquatic exosystems. Mean Cd levels in plants were generally 0.28–0.72 $\mu g\ g^{-1}$ (dry wt), except at one location supplied with ground water pumped from a Zn mine, where plant Cd levels were 10.46 $\mu g\ g^{-1}$. The level of Cd in sediments immediately below the mine was 27.3 $\mu g\ g^{-1}$, compared with 0.04 $\mu g\ g^{-1}$ above the mine. Cadmium levels in muskrat liver and kidney from the contaminated site were 0.32 and 1.07 $\mu g\ g^{-1}$ (wet wt), respectively, while Cd levels in muskrat from the other areas ranged from 0.04–0.14 $\mu g\ g^{-1}$ in liver and 0.17–0.63 $\mu g\ g^{-1}$ in kidney. Radvanyi and Shaw (1980), however, found that metal levels in muskrat tissue did not reflect differences in environmental metal loading near a smelter complex in Canada.

Several researchers have utilized deer as indicators of environmental metal loading. Woolf *et al.* (1982) state that white-tailed deer (*Odocoileus virginianus*) are well suited as biological monitors for broad surveillance because they are widely distributed, abundant, and easily obtained during the annual legal hunting season. From a study of 190 deer representing 15 counties in Illinois (U.S.A.), the authors found a very wide range in the liver concentrations of Cu, Mg, Ni and Zn. Tissue levels of 5 elements (Cr, Cu, Mg, Mn, Pb) could be grouped according to differences between regions, but no explanation for the regional differences were given or postulated. Cadmium concentrations increased with animal age, while Mn levels decreased, and the levels of Cr, Cu, and Mg were greater in males than in females.

King *et al.* (1984) measured Mo and Cu levels in liver tissue of white-tailed deer in Texas from areas around uranium mines and from unmined areas. Molybdenum toxicity (molybdemosis) had been reported in cattle grazing near the mines, and deer from these areas apparently had a high proportion of malformed antlers. However, Mo was detected in only one sample, and based on the study results, the investigators suggest that the animal health problems may be related more to Cu deficiency than Mo toxicity. Krynski *et al.* (1982) reported that tissues (muscle, liver, kidney) of roe deer were suitable indicators of environmental loading of Hg between Hg-treated fields and non-treated forest areas in Poland. While differences in Hg levels between sexes were noted, males and females were collected at different times from the two locations so it is not possible to isolate the relative influence of sex on Hg uptake in this study.

Although fluoride is not a metal, it is an environmental contaminant associated with the manufacture of some metals, most notably Al. Fluorosis involves degeneration of bony tissues in animals, particularly the teeth. Karstad (1967) compared the levels of fluoride in the mandibles and antlers of sick or dying white-tailed deer in the vicinity of an industrial complex in Ontario (Canada) with levels in deer presumed to be normal. The mean fluoride level in mandibles and antlers from the industrial area were 3305 $\mu g\ g^{-1}$ and 1296 $\mu g\ g^{-1}$, respectively, while fluoride levels in mandibles and antlers of healthy deer were 408 $\mu g\ g^{-1}$ and 143 $\mu g\ g^{-1}$, respectively. The level of fluoride in local vegetation was 35.8 $\mu g\ g^{-1}$, and was 1000–1200 $\mu g\ l^{-1}$ in a nearby pond receiving effluent from the plant. Shupe *et al.* (1979) report that the fluoride tolerance in domestic ruminants is about 60 $\mu g\ g^{-1}$ in food and 4–8 $\mu g\ l^{-1}$in water.

Elevated fluoride levels have also been reported in bones of foxes (*Vulpes vulpes*) living near an Al reduction plant in Wales (Walton, 1984). The mean background fluoride concentrations in foxes from several different areas ranged from 293–297 $\mu g\ g^{-1}$, while the mean concentration in foxes collected within 5 km of the plant was 1650 $\mu g\ g^{-1}$. Bone fluoride concentration and fox age were positively correlated.

Mankovska (1980a) found that levels of As, Cd, Pb, and Zn were higher in hair of roe deer in the vicinity of an Al factory and a power plant in Chzechoslovakia compared with a control area. Hair F levels were elevated near the Al plant but not the power plant. The author suggests that analysis of deer hair can be used as an accurate indicator of environmental contamination. Mankovska (1980b) also measured metal levels in the teeth of deer collected near the Al plant. The concentration of F was extremely elevated relative to levels from control areas, while As and Cd levels were moderately elevated. Tooth Pb concentrations were not elevated above control levels.

The Niepolomice Forest in Poland has been subjected to considerable atmospheric deposition of pollutants from several large steel and industrial complexes for many years. Godzinski and Yorks (1981) observed that increased levels of metals (Zn, Pb, Cr, Fe) in antlers of roe deer in this region corresponded with other changes in the ecosystem. For example, there was a decrease in the number of species comprising the forest lichen flora, and also changes in the growth characteristics and metal content of Scotch pine. Elevated levels of Zn, Pb, Cd, and Fe were also recorded in carpet mosses relative to control areas.

Researchers have noted that the size and quality of roe deer antlers collected from the Niepolomice Forest declined between 1920 and 1973 (Jop, 1979). Significant changes occurred in the late 1950's, corresponding to the time when a large iron and steel plant began operation between Krakow and the forest. Weight of the antlers decreased by 32%, length of branches by 29%, and general quality of the antlers decreased 20–30%. Other studies have also noted elevated levels of Zn, Pb, Fe, Cu, and Mg in antlers and forage of deer from the Niepolomice Forest compared with levels in an unpolluted forest (Sawicka-Kapusta, 1979; Grodzinska *et al.*, 1983).

Metal levels (Zn, Pb, Fe, Cr, Cd, Mn, Mg) were also higher in antlers of roe deer collected between 1951–1973 than in antlers of deer collected between 1938–1950. These data further suggest that recent industrial development has had a significant effect on pollutant accumulation in wild deer populations (Sawicka-Kapusta, 1979). The author suggests that roe deer antlers are sensitive bioindicators of forest environmental pollution.

Although cows are a domesticated species, they are frequently free-ranging and are, therefore, exposed to local environmental conditions. In this respect, their value as a biological monitor is similar to a truly wild species from the same area. As such, three separate studies utilizing cows as biomonitors of atmospheric metal loading are briefly mentioned here.

Karacic *et al.* (1984) investigated the absorption of Pb in cows in the vicinity of a Pb smelter in Yugoslavia. The researchers determined that blood Pb levels, as well as aminolevulenic acid dehydratase (ALAD) and erythrocyte protoporphyrin (EP) were accurate biological indicators of environmental Pb exposure. These parameters showed a move toward normalization after filtration devices were attached to the smelters, but still showed elevated Pb absorption in comparison with levels in the control group. This indicated continued exposure to residual Pb remaining in the local soils and vegetation.

Gydesen *et al.* (1981) examined metal levels (Cd, Cu, Pb, Zn, Fe, Mn) in bulk deposition, epiphytic cryptograms and kidneys of cattle from different parts of Denmark to investigate regional atmospheric deposition of pollutants. The regional pattern of Pb, Cd and Cu in kidney tissues agreed with the two other indicators. No difference was noted in kidney Zn levels, however, possibly due to efficient homeostatic regulation of body concentrations. In a study of metals in cows from different areas in Australia, Kramer *et al.* (1983) determined that bovine liver and kidney reflected local sources of As, Cd and Se. Cu and Zn liver values fluctuated randomly between sites, and there was no pattern of any metal levels in muscle tissue.

2.2. Compounding biological factors

In addition to the level of exposure of an organism to an environmental contaminant, there are other intrinsic biological variables which can influence metal uptake and retention in a mammal. Prominant among the biological factors are: (1) sex, (2) age, (3) species, (4) tissue sampled, (5) diet type, and (6) season of sample collection.

Some of these modifying factors are highly interrelated. The difference in metal levels between species may primarily be a function of diet in addition to metabolic differences. Higher Hg levels observed in otter compared with mink, for example, are generally attributed to the fact that fish make up a greater portion of an otter's diet compared with mink, wihch prey upon a variety of items. Andrews *et al.* (1984) found that Cd levels were much higher in insectivorous shrews than in herbivorous voles collected from a revegetated mine site. In addition to higher levels of Cd in insects compared with vegetation, the authors postulate that Cd in insects may be in a more biologically available form.

Metal levels can vary within a species when sampled from different locations. This between-site variability can be due to regional dietary differences. Eaton and Farant

(1982) found that hair Hg levels were higher in polar bears (*Ursus maritimus*) from western areas of the Canadian arctic, compared with polar bears from the eastern arctic and Hudson Bay. It was noted that polar bears from the western arctic tended to eat more bearded seals (*Erignathus barbatus*), which contain higher Hg levels than other seal species, than polar bears from the other regions.

Seasonal fluctuations in animal metal levels may reflect seasonal changes in their diet, or diet composition. The mineral composition of many species of plants and grasses can vary between seasons and location, which in turn can affect the mineral balance of grazing animals. Franzmann *et al.* (1975) determined that the composition of moose hair is a good monitor of the environmental availability of several elements. Levels of Ca, K, Mn, Mg, Na, Cu, Zn, Cd, and Pb were highest in late summer to fall and corresponded with the fall peak of moose condition and fluctuations in browse composition. Differences in Cu levels of moose hair between regions in Alaska were also related to differences in Cu content of brŏwse material (Flynn *et al.*, 1977). A significant correlation between hair Cu levels and dietary intake was also reported in red deer (*Cervis elaphus*) in Hungary (Tolgyesi and Bencze, 1970). Seasonal fluctuations in metal levels of soft tissues of animals have not been investigated in any detail.

The relative influence of sex and age on metal accumulation and retention in mammals is not clearly understood. Table II provides a summary of relationships reported between sex and age and metal concentrations reported in wild mammals. Only Cd levels appear to consistently increase with increasing age of an animal. A positive correlation between tissue Cd concentration and age has been observed in cattle and Columbian ground squirrels (Munshower, 1972), gray squirrels (McKinnon *et al.*, 1976), roe and red deer (Hollerer and Coduro, 1977), mule deer and pronghorn antelope (Munshower and Newman, 1979), white-tailed deer (Kocan *et al.*, 1980; Woolf *et al.*, 1982), and moose (Mattsson *et al.*, 1981; Frank *et al.*, 1981).

TABLE II

Summary of relationships between sex and age and levels reported in wild mammals of various metals

Metal	Sex[a]	Age[b]
Hg	M, F, O	O, +
Pb	M, F, O	O, +
Cd	M, F	O, +
Zn		O
As		O
Cu		−, O, +

[a] M = Metal levels in males greater than in females.
F = Metal levels in females greater than in males.
O = No difference in metal levels between males and females

[b] O = No age affect.
\+ = Metal levels increased with age.
− = Metal levels decreased with age.

Data on the influence of sex and age on Hg levels in mammals are conflicting. O'Connor and Nielson (1980) reported that Hg levels were higher in males than females in both mink and otter. Beck (1977) also found that Hg levels were higher in male otters than in female otters, although the difference was significant in only one of six study areas. Kucera (1983), however, found that Hg levels were higher in female mink, while there was no significant difference in Hg levels between sexes in otters. Higher levels of Hg in mature compared to juvenile animals have been reported for gray squirrels and raccoons (Cumbie, 1975a; Jenkins *et al.*, 1980). No differences in Hg levels relative to age were noted in white-tailed deer (Kocan *et al.*, 1980), red and roe deer (Hollerer and Codura 1977) or otter (Anderson-Bledsoe and Scanlon 1983).

Munshower and Newman (1979) reported that liver Pb levels increased with age of pronghorn antelope, and adult shrews were also found to contain higher Pb levels than immature shrews (Quarles *et al.*, 1977). However, no age-dependent trends in Pb levels were noted in studies of gray squirrels, white-tailed deer, red deer, mule deer or moose (McKinnon *et al.*, 1976; Kocan *et al.*, 1980; Munshower and Newman, 1977; Mattsson *et al.*, 1981). Higher Pb levels in females than in males have been reported for several small rodent species (Roberts *et al.*, 1978; Quarles *et al.*, 1977), while Woolf *et al.* (1982) found slightly higher Pb levels in male white-tailed deer compared with female deer.

Woolf *et al.* (1982) reported that liver Cu levels increased with age of white-tailed deer, and that males contained higher Cu levels than females of comparable age. Stelter (1980) found no difference in the concentration of Cu in liver or kidneys of mule deer of different ages, while Munshower and Newman (1979) reported decreasing Cu levels in livers of mule deer with increasing age, but no difference in kidney Cu concentrations. No age-dependant trends in Cu concentrations were noted in liver or kidneys of moose in Sweden (Mattssan *et al.* 1981). No age or sex-dependent trends have been reported for other metals.

Certain tissues, especially liver and kidney, represent target organs for metals such as Pb, Cd, and Hg. Within an animal, therefore, the concentration of a particular metal will be highest within these target organs. For example, hair generally contains the highest Hg levels of any tissue (Cumbie, 1975b; Wobeser *et al.*, 1976; Sheffy and St. Amant, 1982). Within soft tissues, highest Hg levels are found in liver, followed by kidney and then lesser levels in muscle and brain (Frank *et al.*, 1979; Wren *et al.*, 1980). Mercury levels between different tissues within animals of the same species are often highly correlated (Cumbie, 1975a; Beck, 1977; Sheffy, 1977; Kucera, 1983). Kidney represents the principal target organ for Cd accumulation in mammals, while Pb is primarily deposited in bone tissue. It is important, therefore, that metal levels be compared between similar tissues.

Interactions between vitamins, minerals and other dietary factors also play an important role in metal metabolism, uptake and toxicity within organisms. High Ca levels in drinking water is thought to be responsible for lower hair Zn levels in some populations of children (Gibson *et al.*, 1983), while lowered dietary Ca levels are known to increase the uptake and toxicity of Pb to several animal species (Mahaffey, 1974). Deficient intake of some essential nutrients may increase the rate of absorption of Cd in mammals (Fox, 1979). As most research into the role of nutrition on metal uptake and toxicity has been conducted on laboratory or domestic animal species, the effect of these interactions on metal uptake in wild mammals are generally unknown. However, that nutritional status does effect metal accumulation is well established (Calabrese, 1980), and anyone undertaking biomonitoring studies with wild mammals should be aware of these processes and interactions.

3. Evaluation of Mammals as Biomonitors

One of the principal objectives of this review is to attempt to impartially assess the value of mammals as biomonitors of environmental metal loading. To achieve this aim the merit of mammals used as biomonitors from individual studies was assigned a ranking of poor, good or excellent (Table I). The ranking is based upon examination and interpretation of the study results, as well as discussion or comments of the researchers. Some authors advocate the use of a particular species as a biological monitor (i.e., Mankovska, 1980; Woolf *et al.*, 1982; Kucera, 1983; Cumbie and Jenkins, 1975), while others recommend against the use of an individual species (Beardsley *et al.*, 1978).

The data in Table I is presented by species, tissue and metal. This is necessary to avoid making generalizations and over-simplifications which may not always be appropriate. For example, in his review of biological monitors, Jenkins (1980) rates mammalian liver as an excellent biological monitor of environmental Hg levels. Examination of the literature, however, reveals that while liver tissue of carnivorous or piscivorous mammals is a suitable indicators of Hg levels, the tissues of herbivorous species are generally poor indicators of ambient Hg concentrations. In contrast, liver tissue of some herbivores such as moose and deer appear to be good monitors of environmental levels of Pb and Cd.

Small mammal species have frequently been used as biological monitors of metals in the environment since they meet the criteria of being abundant, are easily caught, do not migrate long distances, have a widespread distribution and appear to accurately reflect environmental concentrations of some metals. A whole-body homogenate of small mammals is adequate for monitoring Pb and Cd, but may not be suitable for other elements such as Ni, Cu, or Zn. The selection of a biological monitoring species for proposed studies must be based upon the objectives, priorities and species availability for each individual study. A whole-body homogenate of rodents would be sufficient, for example, if a study objective was to assess metal levels in the diet of a predatory species feeding on small mammals.

Based on their review of the subject material, Martin and Caughtrey (1982) concluded that the potential of most terrestrial vertebrate animals as biomonitors is limited. One of their major concerns was the sacrifice of animals in order to measure tissue metal concentrations. This is certainly a valid consideration but is a problem that a properly designed study can minimize. For example, full use should be made of animals that are being killed for purposes other then scientific study. A number of mammal species (i.e. rats, beavers, mice, voles) are harvested as a means of pest control, and carcasses of furbearing animals (mink, otter, raccoon) can be obtained from licensed trappers in areas of interest. These individuals are often knowledgable in the natural biology of wild animals, and can provide a great deal of practical information. Similarly, tissues of large game animals (deer, moose) may be obtained through cooperation with hunters during regular hunting seasons. The time limits imposed by regulated hunting and trapping seasons are advantagous in that all animals will be sampled at the same time of year. The principal limitation is that information on seasonal trends in tissue metal levels is then not available.

It is apparent that some essential elements including Zn and Cu do not bioaccumulate in mammal tissue in proportion to environmental levels as do some non-essential elements such as Cd, Pb, or Hg. Effective regulation of the essential elements by an organism will, therefore, tend to depress body burdens of some metals relative to environmental loading. Measurement of these elements within wild species is still important, however, since homeostatic regulation is a true biological response to the ambient situation and sound data is being provided on the ecological fate of a particular contaminant. As Roberts and Johnson (1978) state, the biological implications of metal pollution are not necessarily related directly to the abundance of each contaminant, but rather to their transfer potential and mobility within terrestrial ecosystems.

4. Conclusion

To fully assess the transfer potential of a particular contaminant it is necessary to monitor various physical, chemical and biological compartments of an ecosystem. The deposition of pollutants to, and fate within the environment are related to many physical and biological processes and factors. Thus, the use of integrating methods (i.e. biological monitoring and air dispersion models) is a logical approach to monitoring pollution (Jokinen *et al.*, 1984). Grozinski and Yorks (1981) state that most biological impact studies concentrate on single species with only implicit recognition of the importance of whole system dynamics. They further suggest that it is important to search for ecosystem level processes which might be monitored to delineate likely long-term pollutant effects on the system as a whole.

In some instances, metal levels in mammal tissues may not accurately reflect ambient loading conditions due to both effective homeostatic regulation of tissue concentrations within the organism, as well as differences in environmental mobility between elements. Even in these cases, however, animal tissue levels can provide important data regarding the fate and bioavailability of contaminants within natural ecosystems.

Within this framework, biological monitoring of metal concentrations in wild mammals can play an important role in ecosystem studies. The use of mammals in long-term studies and to complement other environmental studies should be encouraged.

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