MINING-RELATED TRACE ELEMENTS IN RIPARIAN FOOD WEBS OF THE UPPER CLARK FORK RIVER BASIN

Robert A. Pastorok, Andrea J. La Tier, Matthew K. Butcher, and Thomas C. Ginn

Abstract: Fluvial deposition of tailings and other mining-related waste in selected riparian habitats of the Upper Clark Fork River Basin (Montana) have enriched trace elements in soils. The significance of trace element exposure to selected wildlife species was evaluated by measuring tissue residues of trace elements (As, Cd, Cu, Pb, Zn) in key dietary species, including dominant grasses (tufted hairgrass and redtop), willows, alfalfa, barley, invertebrates (grasshoppers, spiders, and beetles), and deer mice. Average trace element concentrations in grasses, invertebrates, and deer mice collected from tailings-affected sites were elevated relative to reference levels. Soil-tissue bioconcentration factors for grasses and invertebrates were generally lower than expected from the range of values in the literature, indicating the reduced bioavailability of trace elements from mining waste. In general, trace elements concentrations in willows, alfalfa, and barley were not elevated above reference levels. From these data and plausible assumptions for other exposure parameters for white-tailed deer, red fox, and American kestrel, trace element intake was estimated for soil and diet ingestion pathways. Comparisons of exposure estimates with toxicity reference values indicated that the elevated concentrations of trace elements in key food web species do not pose a significant risk to wildlife.

Additional Key Words: trace elements, food webs, bioaccumulation.

Proceedings America Society of Mining and Reclamation, 1995 pp 31-51

DOI: 10.21000/JASMR95010051

https://doi.org/10.21000/JASMR95010031

Introduction

The bioavailability of trace elements at mining sites is a key issue in both human and ecological risk assessments (Davis et al. 1992; McKinney and Rogers 1992; Pascoe et al. 1994). Trace elements such as As, Cu, Pb, and Zn may be enriched in soils, sediments, and surface waters as a result of mining-related activities (e.g., mine tailings, smelter emissions). Nevertheless, the degree to which these trace elements are assimilated by biota and transferred through food webs depends on the mineralogy and geochemistry of the parent waste material and the affected abiotic media. Any constraints on bioavailability would in turn limit the toxicity of environmental media affected by the waste material. Thus, an important step in evaluating risks to humans and wildlife is to assess the bioavailability of trace elements by measuring their concentrations in tissues of the exposed receptors or of the primary species in food webs supporting those receptors.

The Clark Fork River Superfund site, located within the Upper Clark Fork River Basin (UCFRB) in western Montana (Fig. 1), encompasses a variety of habitats with elevated concentrations of trace elements (e.g., Johnson and Schmidt 1988; Pascoe and Blanchet 1993). In particular, riverine systems and associated riparian habitats have received mine tailings and other wastes related to past mining practices throughout the region. Studies that addressed bioaccumulation of trace elements have been limited, especially for terrestrial biota (e.g., Munshower 1972, 1977; Pascoe et al. 1994).

A regional ecological risk investigation was conducted within the Clark Fork River Superfund site to evaluate 1) the relationships between vegetation communities and tailings deposits in riparian habitats and 2) food-chain transfer of trace elements to selected wildlife species (ARCO 1994). The results of the vegetation community studies are reported in a companion paper (Pastorok et al., in press). This report focuses on the evaluation of risk to wildlife based on the measurement of bioaccumulation of trace elements in native or naturalized plants, terrestrial invertebrates, and the deer mouse (*Peromyscus maniculatus*). For large wildlife species, such as raptors, white-tailed deer (*Odocoileus virginianus*), and red fox (*Vulpes vulpes*), bioaccumulation data were interpreted in the context of food web exposure models. Because direct assessment of toxicological effects in small mammals is practical, potential reproductive effects in the deer mouse were evaluated by direct measurements (reported in a separate paper by La Tier et al., in press).

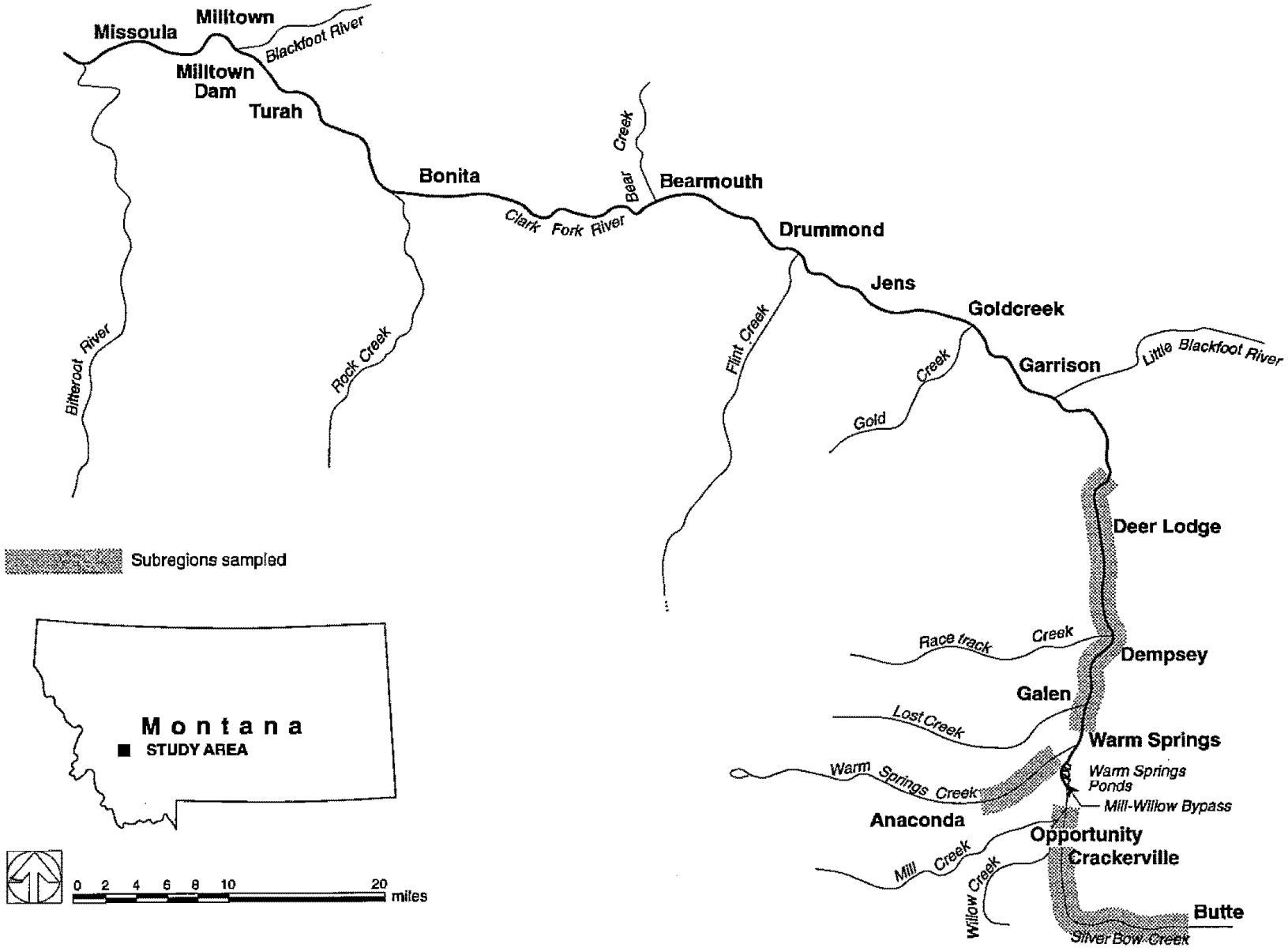

Figure 1. Upper Clark Fork River Basin subregions.

The focus of this investigation was on riparian habitats and adjacent transitional areas (i.e., dry areas adjacent to riparian habitat and not within floodplains). Because physical-chemical conditions and biological communities differ greatly among habitat types, these results should not be extrapolated to other habitats within the UCFRB (e.g., upland areas) or to other regions.

Methods

Endpoints for this study are summarized in Table 1. PTI (1993) and ARCO (1994) describe sampling and analysis methods and quality assurance procedures in detail. EPA-approved or EPA-recommended analytical methods were used whenever possible. Modifications and exceptions to the EPA methods are described below. Aside from the additional quality assurance procedures required for this specific project, all quality assurance procedures were in accordance with the Clark Fork River Superfund site investigations quality assurance project plan (ARCO 1992).

Stations

Sampling stations are shown in Figures 2 and 3 and described in detail in ARCO (1994). Samples were collected from riparian and transitional habitats in each of three subregions of the Clark Fork River Superfund site: Silver Bow Creek, Warm Springs Creek, and the Upper Clark Fork River. Characteristics of these subregions are described in ARCO (1994).

Vegetation samples were collected only from riparian areas because of the concentration of mine tailings deposits in these areas. For the small mammal and invertebrate components, both riparian and transitional areas were sampled. The transitional areas were sampled to evaluate the spatial distribution of mice with elevated body burdens of trace elements because individuals caught in transitional areas could have traveled to riparian areas where they might have been exposed to elevated concentrations of trace elements in soil, invertebrates, and vegetation.

Selection of Species

For the evaluation of risk to wildlife, bioaccumulation of trace elements was evaluated in key food resources for the wildlife species selected for a screening risk assessment (ARCO 1994). The selected receptors are white-tailed deer, red fox, and American kestrel (*Falco sparverius*). The primary food resources of these receptors include dominant plant species (e.g., grasses, forbs, and willows), terrestrial invertebrates (i.e., beetles, grasshoppers, and spiders), and the deer mouse. Most of the species selected for measurement of trace element concentrations are common in riparian habitats across all subregions of the UCFRB. Although variations in local abundance of invertebrates required that different species be selected in different subregions, the same species groups (i.e., beetles, grasshoppers, and spiders) were collected at all stations.

Sampling and Analysis

Riparian Plants—For analysis of trace elements in vegetation, samples of grasses were collected at 22 stations distributed among the three subregions and at six reference stations (Fig. 2). The species included tufted hairgrass (*Deschampsia caespitosa*), redtop (*Agrostis alba*), slender wheatgrass (*Agropyron trachycaulum*), and Kentucky bluegrass (*Poa pratensis*). Aboveground plant parts along the perimeter of the 10 quadrats used for measurements of plant biomass and soil chemistry (Pastorok et al., in press) were clipped and combined in one composite sample for trace element analysis.

Willow leaves were collected at four stations along Silver Bow Creek and at one reference site in Browns Gulch (Fig. 2). Five individual samples of leaves were clipped and combined in one composite sample for trace element analysis.

Table 1. Endpoints for the regional ecorisk field investigation

Study Component	Area of Investigation	Number of Species	Chemical Endpoint	Biological Endpoint
Soil Chemistry[a]	SBC, WSC, UCFR, 3 reference areas	NA	Total As, Cd, Cu, Pb, Zn; ancillary variables: moisture, percent solids, chloride, grain size, cation exchange capacity, total organic carbon, sulfate, pH, soluble phosphorus, total Kjeldahl nitrogen	NA
Plant Community	SBC, WSC, UCFR, 3 reference areas	>1	NA	Cover, species diversity, biomass production
Plant Bioaccumulation	SBC, WSC, UCFR, 1 reference area	3[b]	Total As, Cd, Cu, Pb, Zn; arsenic speciation (WSC subregion only); percent solids	NA
Crop Bioaccumulation	SBC, WSC, UCFR, 1 reference area	1	Total As, Cd, Cu, Pb, Zn; percent solids	NA
Invertebrate Bioaccumulation	SBC, WSC, 1 reference area	3[c]	Total As, Cd, Cu, Pb, Zn; percent solids	NA
Small Mammal Bioaccumulation	SBC, WSC, UCFR, 1 reference area	1	Total As, Cd, Cu, Pb, Zn; percent solids	NA
Small Mammal Reproductive Effects	SBC, WSC, UCFR, 1 reference area	1	NA	Bands on uteri, implantation sites, embryos; field measures include age class, sex, weight, testicular status, vaginal status, lactation status

Note: NA - not applicable
SBC - Silver Bow Creek
UCFR - Upper Clark Fork River
WSC - Warm Springs Creek

[a] During the reconnaissance trip, other soil parameters were measured in the field to ensure comparability of sites.

[b] In each subregion, three species were selected for analysis of trace elements in above-ground plant tissues. At some stations, more than one species was sampled.

[c] Three species/species groups were selected at each station. Efforts were made to select the same species across stations.

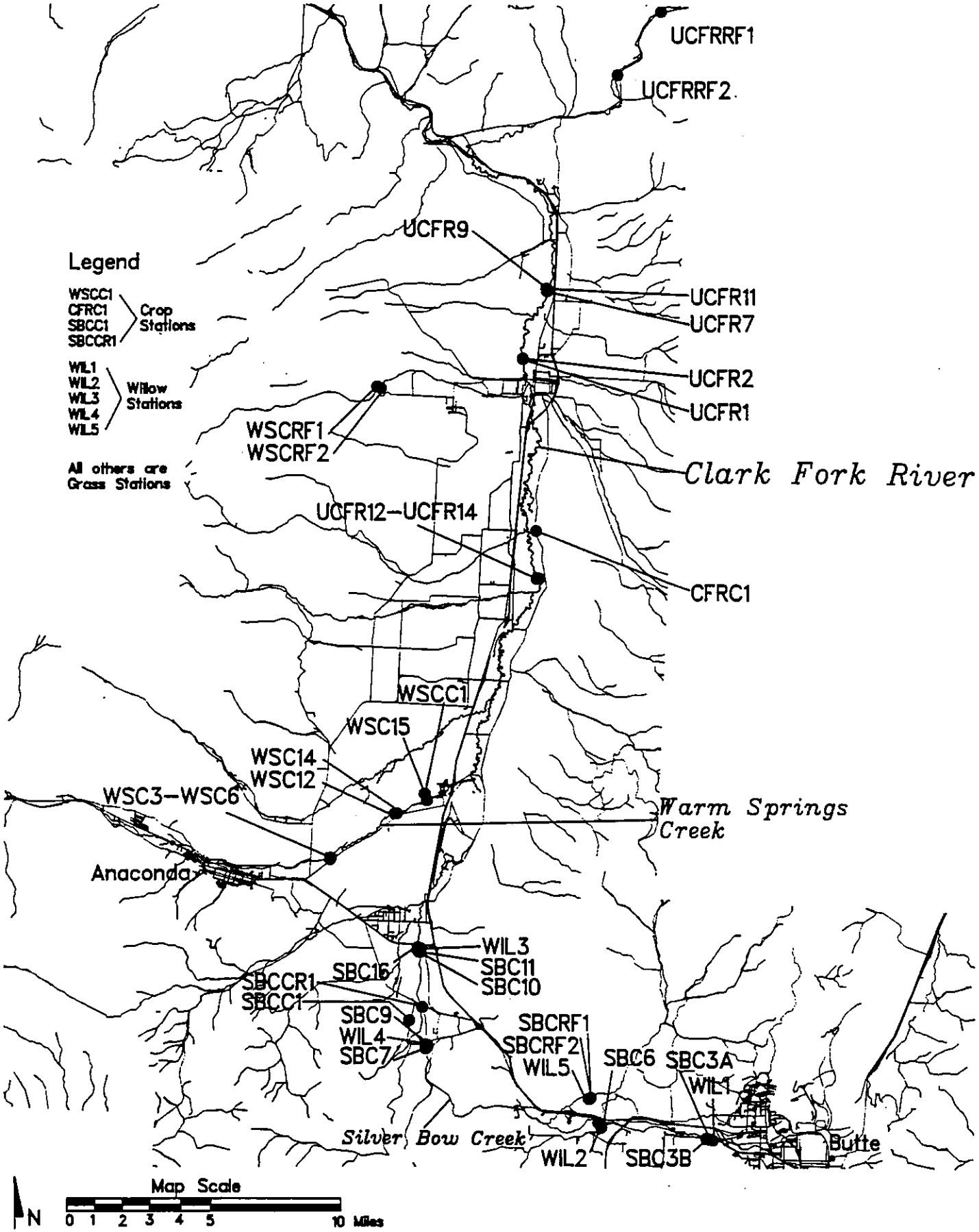

Figure 2. Riparian plant sampling stations for bioaccumulation analyses.

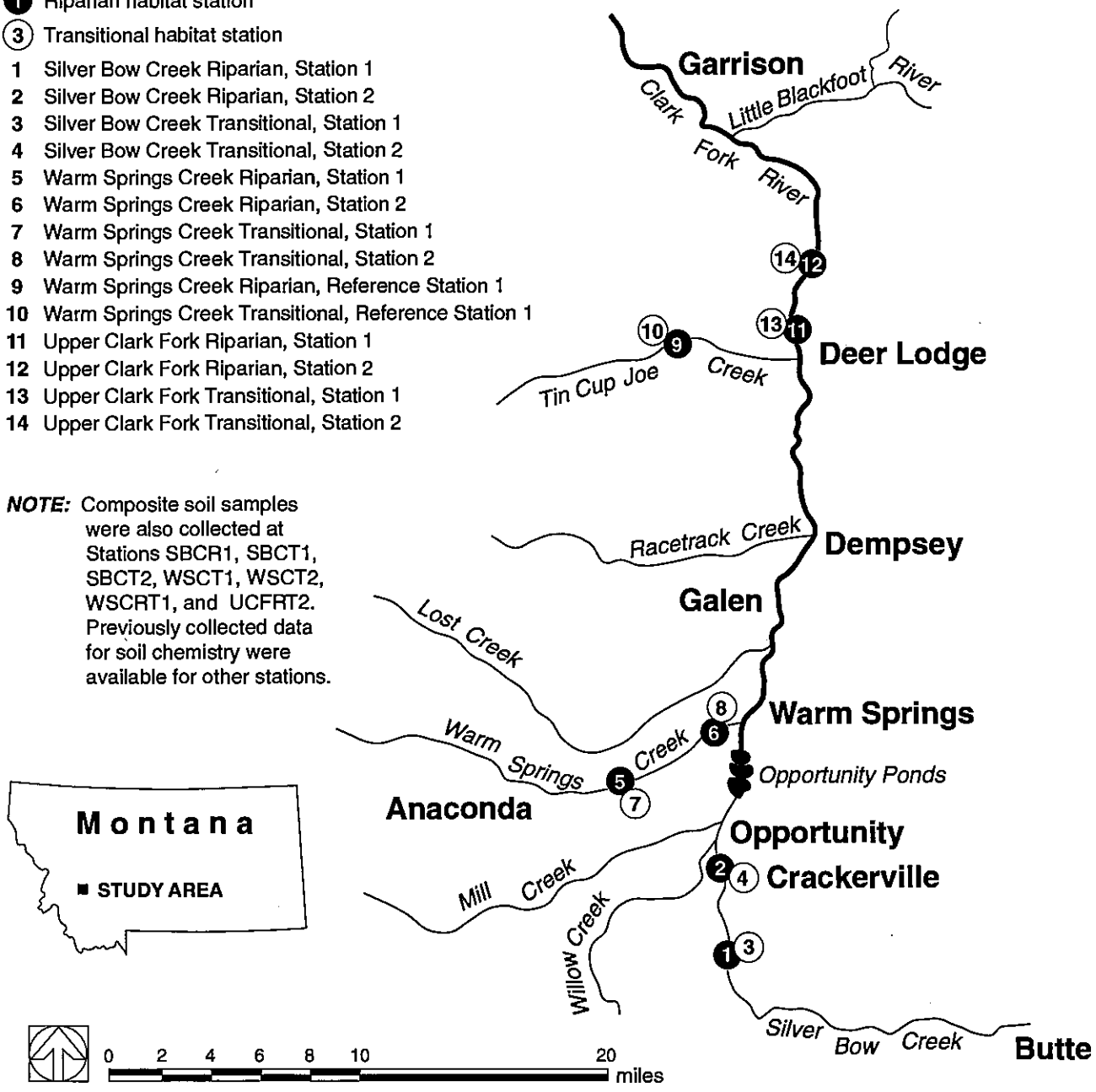

Figure 3. Mammal and invertebrate sampling stations.

Plant tissue samples were analyzed for target trace elements with the Contract Laboratory Program (CLP) statement of work (SOW) 788 (U.S. EPA 1988), except tissue was digested for trace elements by modified EPA SW-846 Method 3050, with an additional quantity of acid.

Agricultural Plants—Samples of alfalfa (*Medicago lupulina*) and barley (*Hordeum* spp.) were collected from several stations (Fig. 2) by methods similar to those used for collection of grasses. Analytical methods were similar to those used for riparian plants.

Invertebrates—Invertebrate samples were collected with pit traps installed at four corners of each selected small mammal trapping grid (Fig. 3), cover boards, and sweep nets. Composite samples of beetles, grasshoppers, and spiders were collected at each site. The samples were analyzed for trace elements by CLP SOW 788 (U.S. EPA 1988), with digestion by modified EPA SW-846 Method 3050 with an additional quantity of acid.

Small Mammals—At each small mammal station (Fig. 3), four grids consisting of a total of 200 Sherman live traps were sampled (see ARCO 1994 for details). The first three adult deer mice (weighing 10 g or more) trapped per grid were sacrificed and placed in a composite sample for the station (12 individuals total). Exceptions to this procedure were made when trap success was low.

Laboratory analyses were conducted in accordance with CLP SOW 788 (U.S. EPA 1988). The mammal tissue samples were prepared according to a modified Montana State laboratory procedure in which the samples were homogenized, freeze-dried, then digested with concentrated nitric acid (CAS 1990).

Data Analysis

Food web exposure models for red fox, white-tailed deer, and American kestrel were used to interpret bioaccumulation data for key species of plants, invertebrates, and small mammals. Each food web exposure model was applied to estimate the individual exposure expressed as a total daily dose ($IR_{ingestion}$) for the receptor species of concern. Estimation of a site-specific dose ($IR_{ingestion}$) allows direct comparison of exposure estimates to toxicity reference values (TRVs) for terrestrial wildlife species. TRVs are typically reported as the threshold daily dose to an individual. For this study, no-observed-adverse-effects levels (NOAELs) were used as TRVs.

The general structure of the exposure models is as follows:

$$IR_{ingestion} = \sum_h (T_h)(IR_h) = \sum_h (T_h) \left(\sum_i \frac{(C_{ih} \times M_i \times A_i)}{W} \right)$$

where

$IR_{ingestion}$ = species-specific total rate of chemical intake by ingestion in all habitats (mg/kg-day)

T_h = fraction of time that species spends in habitat h

IR_h = species-specific rate of chemical intake by ingestion in habitat h (mg/kg-day)

C_{ih} = concentration of the chemical in medium i in habitat h (mg/kg)

M_i = rate of ingestion of medium i (kg/day)

A_i = relative gastrointestinal absorption efficiency for the chemical in medium i (proportion)

W = body weight of receptor species (kg).

The term IR_h can be expanded to specify each ingested medium:

$$IR_h = \left[\sum(C_{plants} \times M_{plants} \times A_{plants}) + \sum(C_{animals} \times M_{animals} \times A_{animals}) + (C_{soil} \times M_{soil} \times A_{soil}) + (C_{water} \times M_{water} \times A_{water})\right]/W$$

where chemical concentrations in plants, animals, soil, and water may vary among habitats. To account for ingestion of different food types by a given receptor, the intake rates for plants and animals are expressed as summations.

Site-specific data on trace element concentrations in soil and key food species of the receptor of concern were incorporated into the model to estimate chemical intake. For red fox and white-tailed deer, ingestion of soil was included as a potential exposure pathway in the model. Significant quantities of contaminated soil can be consumed by these species through feeding and preening. ARCO (1994) provides details of methods used to calculate mean concentrations of trace elements in soils depending on the amount and spatial distribution of data. Ingestion of surface water was not included.

To account for the relative bioavailability of trace elements in various media, a gut absorption factor was applied. The concentration of a trace element in each medium is multiplied by a medium-specific relative absorption factor (A_i) when calculating $IR_{ingestion}$ for the receptor. The gastrointestinal absorption efficiency equals 1.0 if a given medium is the same as that used in the toxicity study from which the TRV is derived.

Results and Discussion

Trace Element Bioaccumulation

Grasses—Trace element concentrations in aboveground tissue of grass species from the three subregions were higher than those from the reference area (Fig. 4). Trace element concentrations for grasses in study areas ranged from 1.70 to 24.9 mg/kg for As, 0.0690 to 3.70 mg/kg for Cd, 4.20 to 325 mg/kg for Cu, 0.200 to 38.2 mg/kg for Pb, and 39.0 to 999 mg/kg for Zn. The grasses from the Warm Springs Creek subregion had higher concentrations of As than those from the other subregions (Fig. 4). For the other four trace elements, the grasses from the Silver Bow Creek subregion had the highest concentrations. Besides having the highest concentrations of Cu and Pb in grasses, Station SBC3 also had the second or third highest concentrations of As, Cd, and Zn.

Bioconcentration factor (BCF) values for accumulation of trace elements by grasses are shown in Fig. 5. Among subregions, median BCF values were generally higher as trace element concentrations in soil and in plant tissue increased. At very low concentrations of trace elements in the reference areas, BCF values were typically higher. The BCF values measured in this investigation are less than or equal to those reported by PTI (1991) and Kabata-Pendias and Pendias (1992) in their literature reviews (Fig. 6).

Willows—Leaves and stems of the three species of willows (*Salix exigua, S. glauca,* and *S. bebbiana*) that were analyzed did not bioaccumulate trace elements in appreciable amounts, with the exception of Zn (Table 2). Willows collected from each of the four stations along Silver Bow Creek showed slightly elevated concentrations of Zn (approximately 2–4 times the reference value of 245 mg/kg). Willows at Station SBC10 contained the greatest concentration of Zn (1,090 mg/kg). Arsenic was undetected at 1 mg/kg at all stations except WIL2 where it was undetected at 1.4 mg/kg.

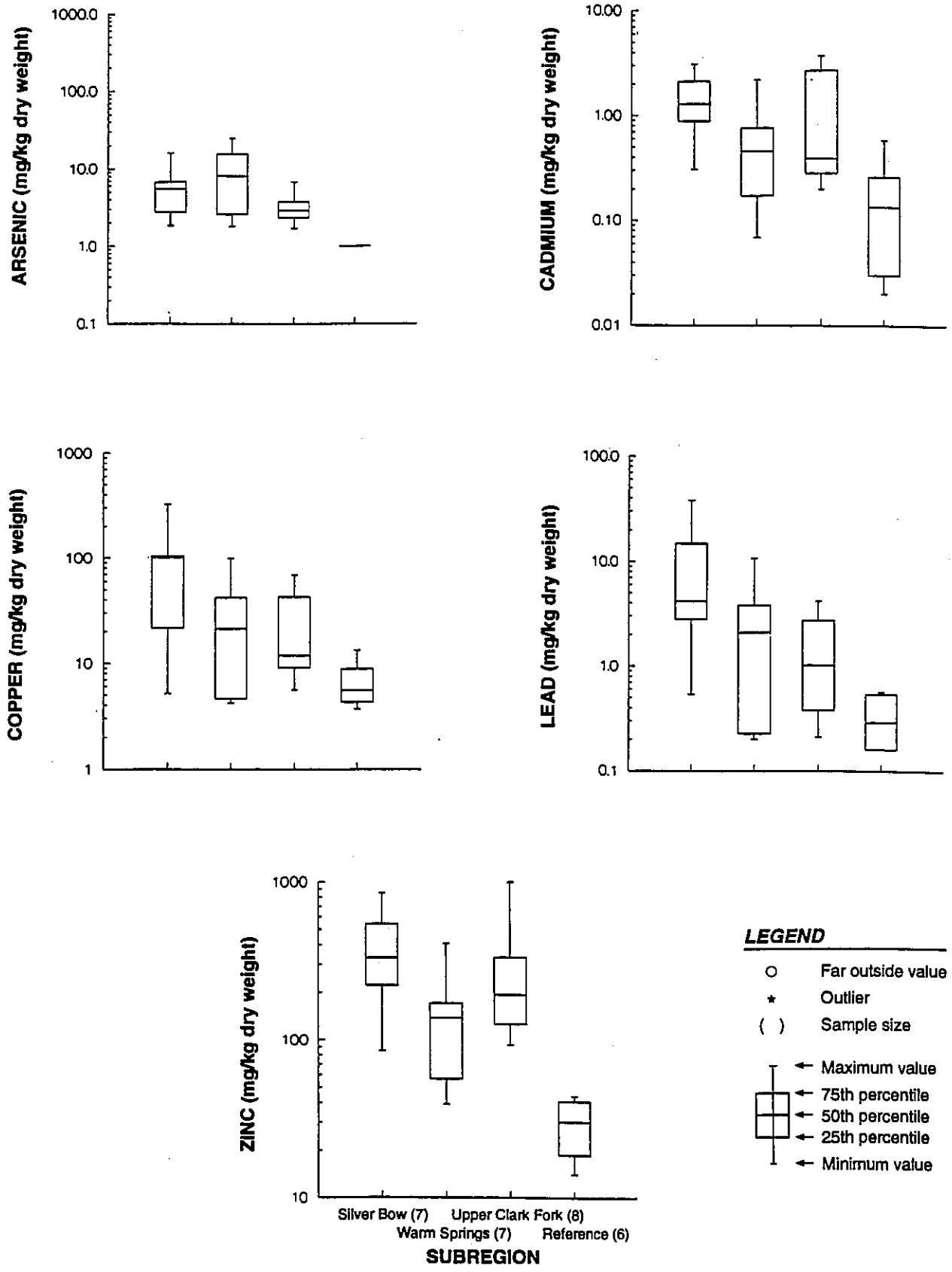

Figure 4. Trace element concentrations in above ground tissue of riparian grasses by subregion.

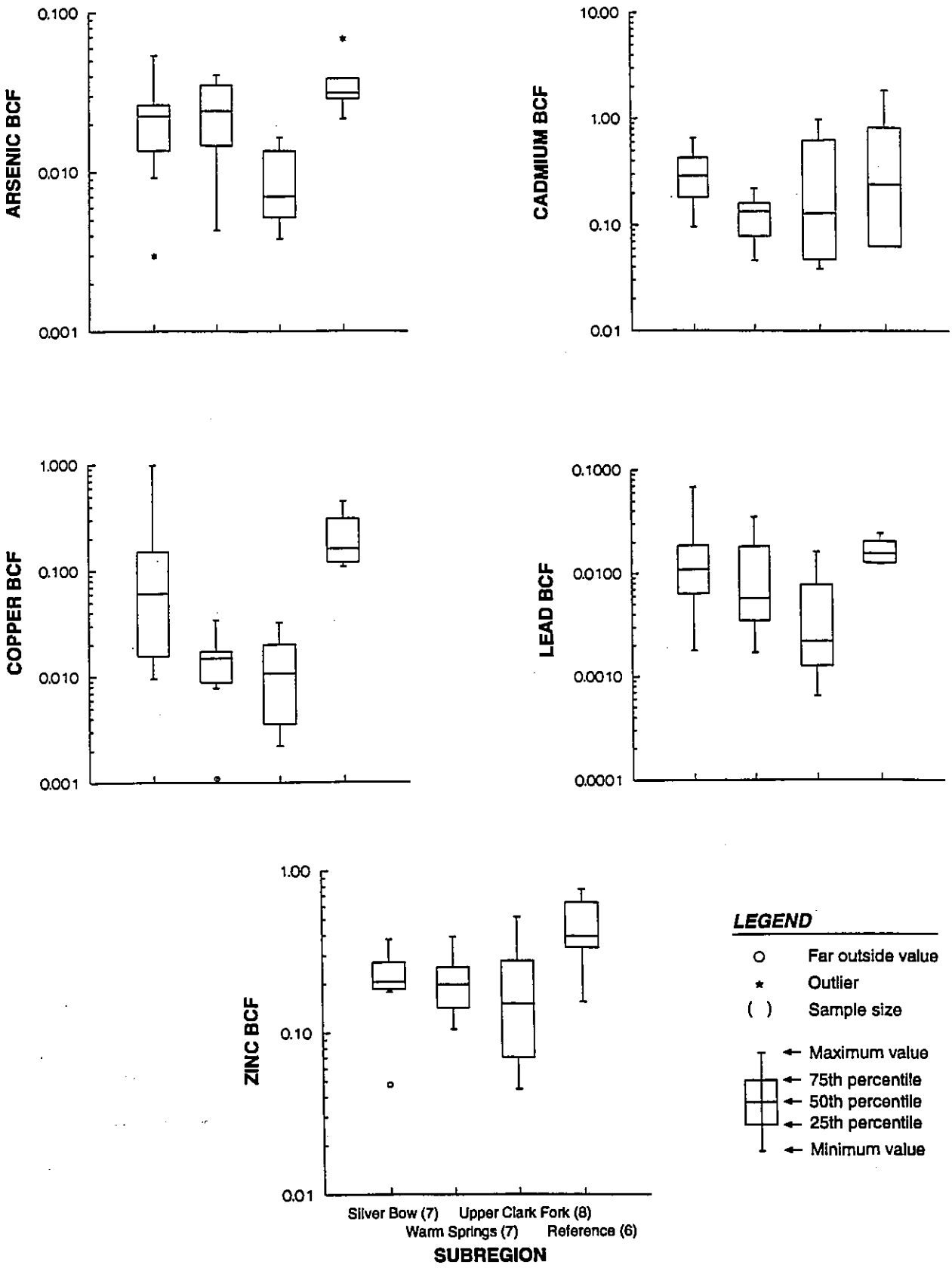

Figure 5. Bioconcentration factors for trace elements in riparian grasses and soil by subregion.

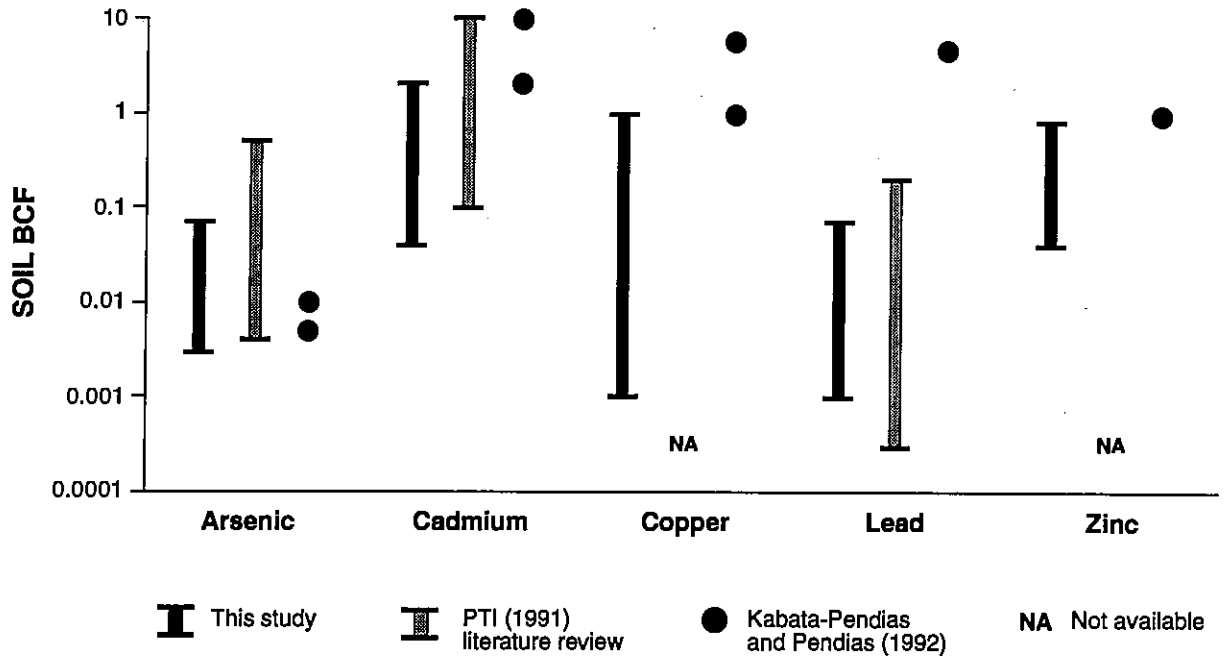

Figure 6. Soil-trace element bioconcentration factors for plants.

Crops—Trace element concentrations in alfalfa and barley were similar among stations, within a factor of 3 (Table 2). Accounting for analytical variance, the trace element concentrations in crops were within the range of reference area data (alfalfa). Arsenic was detected at 2.80 mg/kg in barley at Station WSCC1. Considering typical analytical variability, this value is not considered to be significantly greater than the alfalfa reference value of 1.61 mg/kg. Also, trace element concentrations in crop samples were generally less than or equal to the median trace element concentrations in grasses from reference areas (Fig. 4).

Invertebrates—Trace element concentrations in beetles, grasshoppers, and spiders at selected small mammal trapping stations were elevated compared with reference values (maximum elevations of approximately 10 times for some trace elements; Fig. 7). No differences were apparent between the two subregions sampled. The riparian station at Silver Bow Creek tended to have higher concentrations of trace elements in these terrestrial invertebrates than the corresponding transitional station, whereas the opposite trend was observed at Warm Springs Creek stations. The trend at Silver Bow Creek reflected the relative soil concentrations of all five trace elements in areas where the invertebrates were collected (see ARCO 1994). For Warm Springs Creek, average concentrations of trace elements in soil samples from the transitional area station were not always higher than those at the riparian station (ARCO 1994). Only As and Pb were clearly elevated in the transitional areas compared with the riparian station. Because the invertebrate samples represented species groups and not necessarily individual species, differences between stations may be influenced by species differences in bioaccumulation potential.

Spiders generally exhibited higher body burdens of trace elements than did beetles and grasshoppers (Fig. 7). Beetles and grasshoppers typically had similar body burdens of trace elements, except for Cu where concentrations in grasshoppers were approximately 1.5–2 times those in beetles. Beetles had higher body burdens than grasshoppers at Stations WSCR1 and WSCT1, which had the highest concentrations of As in soil among those sampled for invertebrates (average soil As concentrations of 586 mg/kg and 1,400 mg/kg, respectively).

BCF values for accumulation of trace elements by terrestrial invertebrates are shown in Figure 8. Across all three invertebrate taxa, BCF values for the Silver Bow Creek subregion and the reference area (Tin Cup

Joe Creek) were generally higher than those for the Warm Springs Creek subregion. This trend may result from an increase in BCF values as soil concentrations of trace elements decrease, which was also observed for trace elements bioaccumulation in grasses (see above, *Grasses*). At very low concentrations of trace elements in the reference areas, BCF values were typically higher.

Table 2. Trace element concentrations in willows and agricultural crops

Station	Species	Arsenic (mg/kg DW)	Cadmium (mg/kg DW)	Copper (mg/kg DW)	Lead (mg/kg DW)	Zinc (mg/kg DW)
Willow Study Area						
WIL1	Willow	1.00 *U*	3.50 *JD*	9.90	1.40	909
WIL2	Willow	1.37 *U*	3.67 *JDLX*	11.5	1.56	536
WIL3	Willow	1.00 *U*	4.70 *JD*	10.6	1.10	1,090
WIL4	Willow	1.00 *U*	5.50 *JD*	6.90	0.450	702
	Minimum	1.00	3.50	6.90	0.450	536
	Maximum	1.37	5.50	11.5	1.56	1,090
	Mean	1.09	4.34	9.72	1.13	809
Willow Reference Area						
WIL5	Willow	1.00 *U*	4.70 *JD*	9.40	0.490	245
Crop Study Area						
SBCC1	Alfalfa	1.00 *U*	0.180	11.9	0.130	42.2
CFRC1	Alfalfa	1.03 *U*	0.0653	9.33	0.130	22.7
WSCC1	Barley	2.80	0.0540	9.70	0.300	28.6
	Minimum	1.00	0.0540	9.33	0.130	22.7
	Maximum	2.80	0.180	11.9	0.300	42.2
	Mean	1.61	0.0998	10.3	0.187	31.2
Crop Reference Area						
SBCCR1	Alfalfa	1.30	0.220	10.6	0.590	38.4

Note: For undetected data, the detection limits are included in all calculations.

D - duplicate relative percent difference control limits were exceeded
DW - dry weight
J - estimated
LX - specified frequency of laboratory control sample analysis was not satisfied
U - undetected

Direct comparison of soil-invertebrate BCF values measured in this study with values from other studies is not appropriate because BCF values for terrestrial invertebrates vary greatly with species (Roth 1992; Ma 1994) and different studies use various schemes for compositing across species. Despite this limitation, it should be noted that the soil-spider BCF values found in this study for Cd (BCF=0.53–18) were lower than those reported by Ma (1994) (BCF=18–86), whereas the Pb BCF values in Figure 8 (BCF= 0.07–1.12) encompassed the range reported by Ma (1994) (BCF=0.3–0.5). For beetles, the BCF values found in this study for both Cd (BCF=0.32–2.38) and Pb (BCF=0.02–0.24) were lower than those reported by Ma (1994) (BCF=1.3–9.5 for Cd; BCF=0.09–0.6 for Pb).

Small Mammals—Trace element concentrations in whole deer mice were higher at some study sites than at the reference areas (Fig. 9). In general, deer mice at Warm Springs Creek accumulated more As, Cd, and Cu than animals from other subregions and from the reference area. As body burdens of deer mice

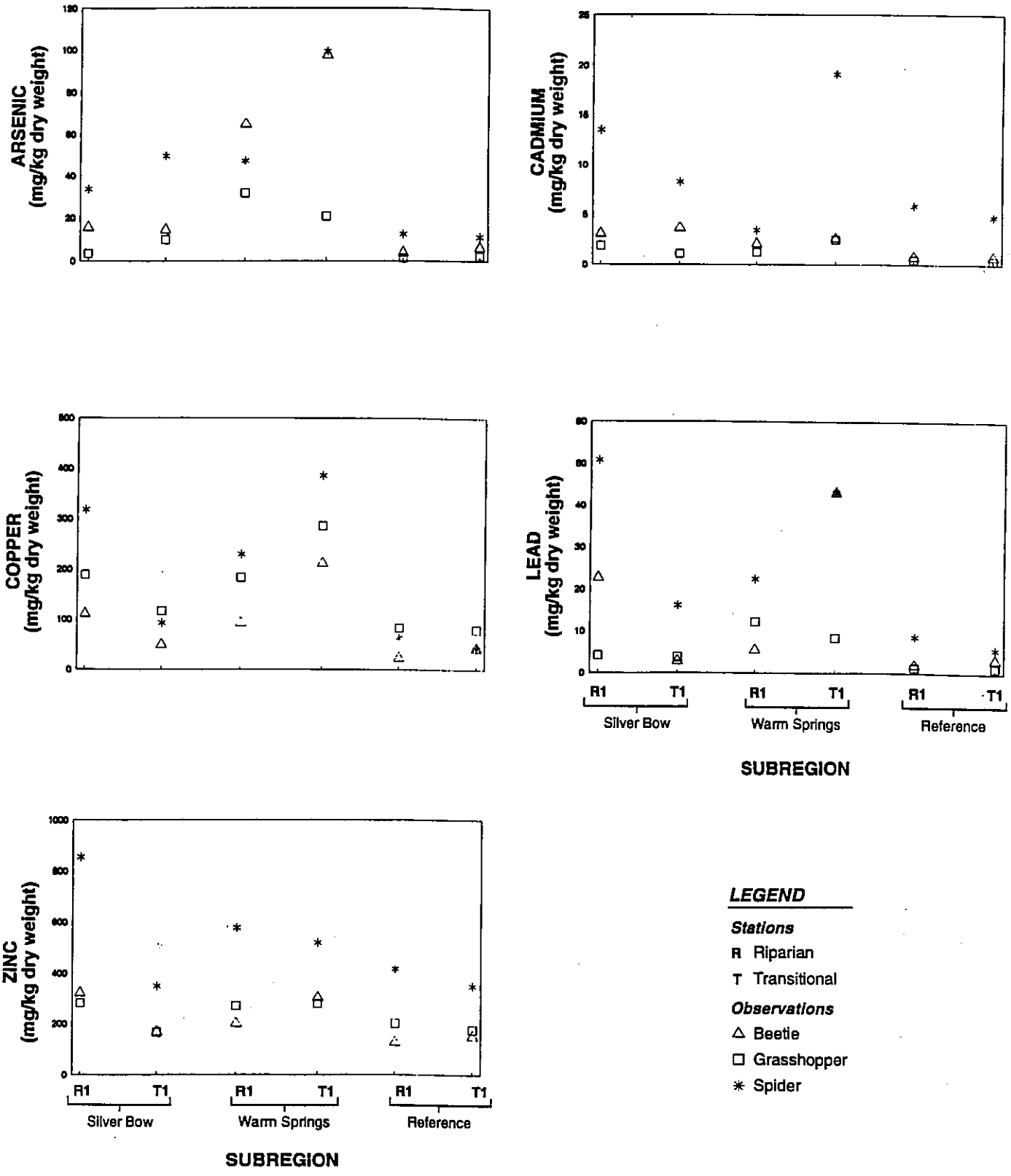

Figure 7. Trace element concentrations in terrestrial invertebrates by station.

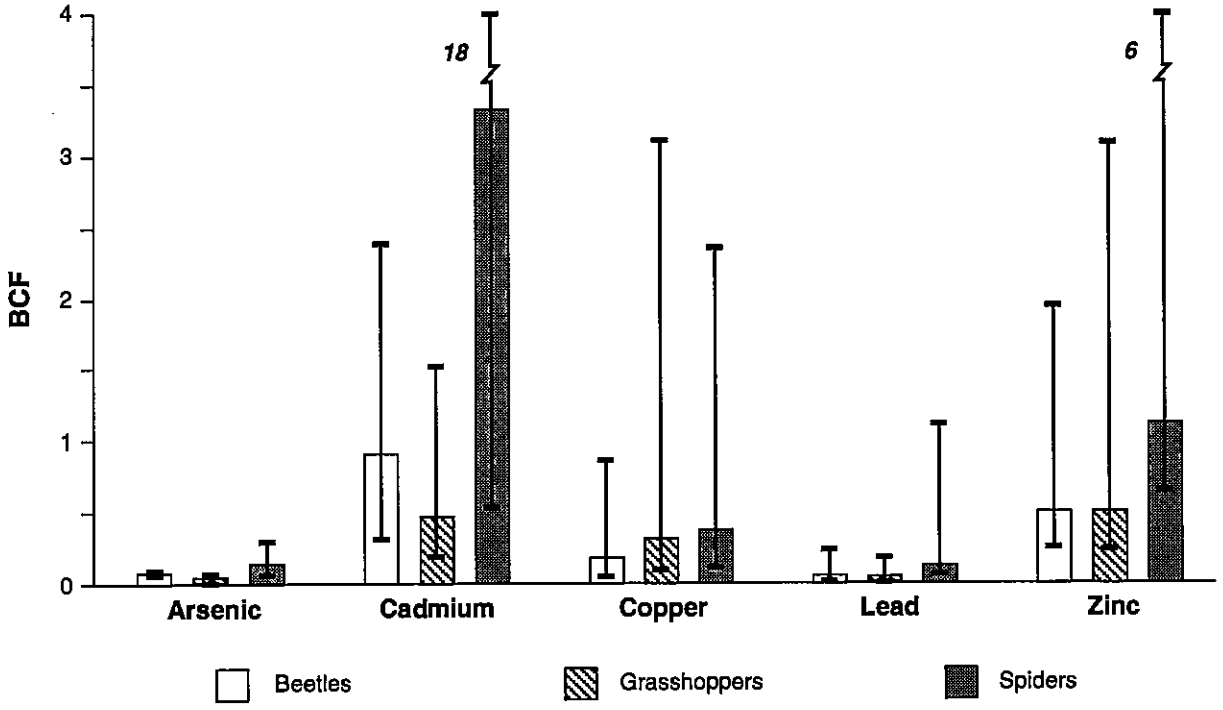

Figure 8. Soil-trace element bioconcentration factors for invertebrates.

collected at Station WSCT1 (11 mg/kg) were approximately 3 times greater than those at the reference area. Cd levels in mice collected at Silver Bow Creek and Warm Springs Creek were elevated relative to reference values, whereas levels in mice collected from the Upper Clark Fork River subregion were not. Even in the former areas, whole-body concentrations of Cd in deer mice were minimal (less than 1.0 mg/kg). Cu concentrations in deer mice were elevated relative to reference values only at selected stations (Stations WSCR1, WSCT1, and UCFRR2). Animals from Station UCFRT1 bioaccumulated the greatest amount of Zn (468 mg/kg), and deer mice from Station SBCT2 accumulated the greatest amount of Pb (10.6 mg/kg). Pb concentrations in mice at all Silver Bow Creek and Warms Springs Creek stations were higher than those at the reference area. In the Upper Clark Fork River subregion, only Station UCFRR2 showed a substantial elevation above reference for Pb.

Overall, As concentrations in whole deer mice, invertebrate tissues (all three functional groups), and soil were highest at transitional habitat stations in the Warm Springs Creek subregion. Because deer mice consume invertebrates and live in close association with soils, they are exposed to As through dietary and soil intake. The diet of deer mice is primarily seeds supplemented by invertebrates. Although vegetation samples for analysis of trace elements were not collected from the mammal stations, evaluation of soils and plant tissue data at other stations demonstrates the potential for bioaccumulation of trace elements in plants. A substantial portion of As taken up by plants can be translocated to the seeds (Kabata-Pendias and Pendias 1992). Thus, both plant material and invertebrates could be dietary sources of As for deer mice.

Cd accumulation in deer mice was low and did not exceed 1.0 mg/kg. It has been shown that Cd is highly mobile and readily bioconcentrates in the terrestrial food chain (Martin and Coughtrey 1976 and Martin et al. 1976, as cited in Roberts and Johnson 1978; Van Hook 1974, as cited in Roberts and Johnson 1979). In mammals, Cd accumulates in the soft tissues with less than 10 percent of the total body burden being stored in the skeleton. The fact that mammals in this study did not accumulate appreciable amounts of Cd may be a result of diet. Although Cd concentrations in grasses and invertebrates sampled in this study were higher than reference values (see above, *Grasses* and *Invertebrates*), the elevations were not great enough to result in a detectable elevation in whole deer mice.

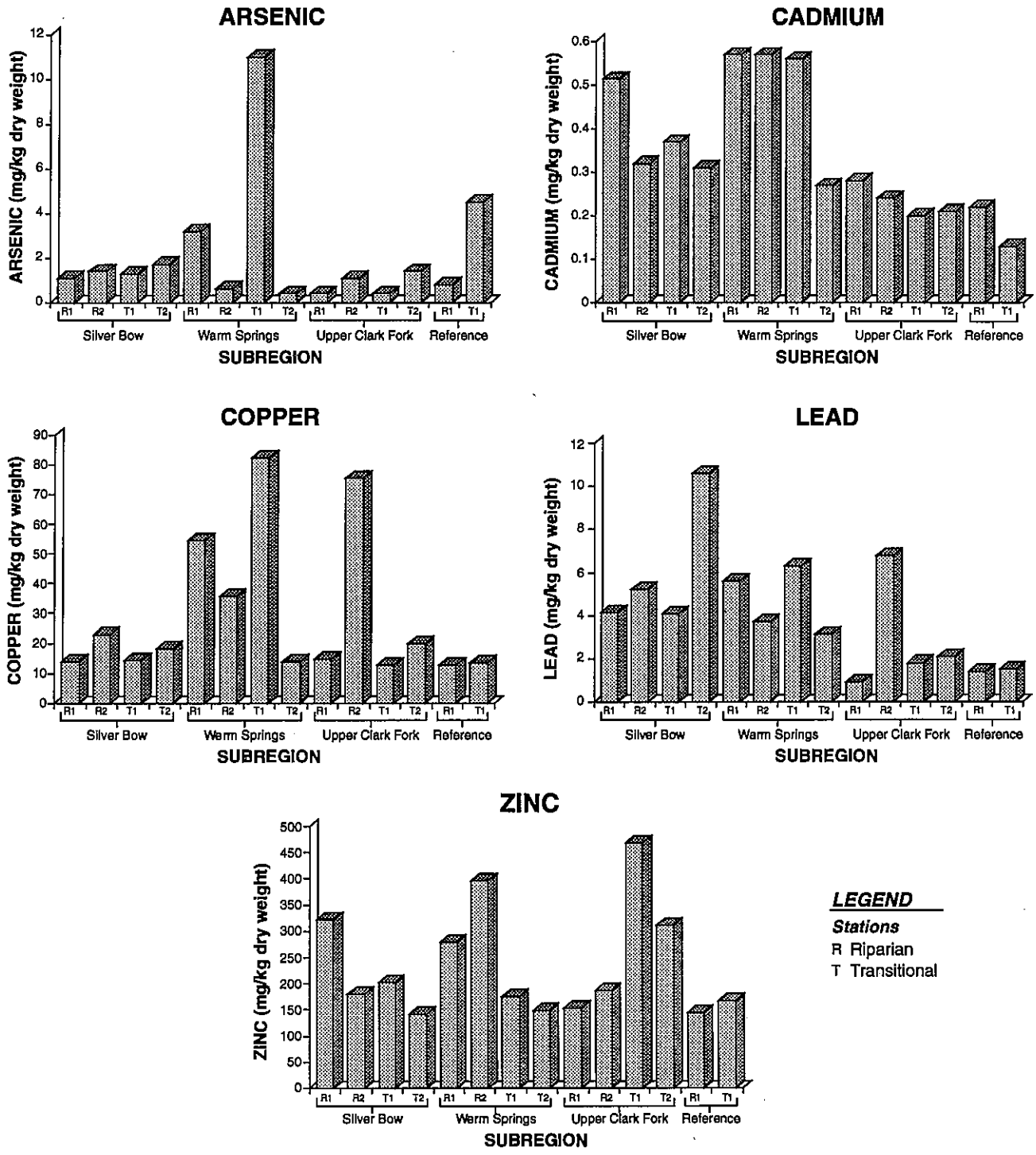

Figure 9. Trace element concentrations in deer mice *(Peromyscus maniculatus)* by station.

Tissue concentrations of Cu and Zn in deer mice from this study are higher than values for *Peromyscus* spp. from other mining/smelting sites (Roberts and Johnson 1978) and from unmined ore deposits (Smith and Rongstad 1982). However, the trace elements in deer mice from the Upper Clark Fork River system did not result in measurable effects on growth or reproduction. Essential elements including Cu and Zn are generally better regulated than nonessential elements such as As, Cd, and Pb. The ability of higher animals to regulate the absorption and retention of Zn is well known and probably reflects the essential role of Zn in biological systems (Vallee 1959, as cited in Roberts and Johnson 1978). Roberts and Johnson (1979) demonstrated that homeostatic mechanisms are relatively unaffected by the high Zn concentrations associated with areas highly contaminated with trace elements.

Pb is not readily passed up the food chain because the majority of the total body burden is deposited in skeletal components. Concentrations of Pb in *Peromyscus* spp. at a mining site sampled by Roberts and Johnson (1978) were higher than those found in this study.

The BCFs for trace elements in deer mice and soils were generally highest for the riparian and transitional habitat at the reference area and the transitional habitats along the Upper Clark Fork River (Table 3). Overall, trace element concentrations in soil were lowest in these areas. Tissue-soil BCFs were generally highest for Zn and Cd (Table 3). At another mining site, Johnson et al. (1978) showed that Zn and Cd are bioaccumulated more readily than Pb in omnivorous small mammals.

Table 3. Soil-trace element bioconcentration factors for deer mice

	Bioconcentration Factors (BCFs)				
Station	Arsenic	Cadmium	Copper	Lead	Zinc
SBCR1	0.01	0.18	0.03	0.01	0.27
SBCR2	0.01	0.10	0.03	0.01	0.16
SBCT1	0.01	0.17	0.07	0.06	1.37
SBCT2	0.01	0.03	0.02	0.01	0.07
WSCR1	0.01	0.09	0.03	0.02	0.35
WSCR2	0.005	0.14	0.19	0.03	2.51
WSCT1	0.01	0.12	0.07	0.01	0.22
WSCT2	0.003	0.08	0.04	0.03	0.46
UCFRR1	0.001	0.05	0.01	0.004	0.11
UCFRR2	0.003	0.04	0.03	0.02	0.11
UCFRT1	0.01	0.20	0.18	0.06	5.85
UCFRT2	0.04	0.28	0.27	0.09	3.78
WSCRFR1	0.02	0.69	0.47	0.18	2.20
WSCRFT1	0.07	0.08	0.11	0.04	1.18
Mean BCF[a]	0.01	0.12	0.08	0.03	1.27

[a] Reference area data (Stations WSCRFR1 AND WSCRFT1) were not included in calculation of BCFs.

Tissue-soil BCF values calculated for deer mice in this study were compared with similar whole-body BCF values reported in the literature for small mammals. The BCF value reported by Johnson et al. (1978) for Cd in *Apodemus sylvaticus* was 0.28, which is equal to the maximum BCF value for Cd in deer mice measured during this study. *A. sylvaticus* is an omnivorous rodent that is the ecological equivalent of *Peromyscus maniculatus* in North America. Beyer et al. (1985) measured Cd in a white-footed mouse (*P. leucopus)* carcass that resulted in a BCF value of 0.008 for Cd. A BCF value of 0.04 was reported by Beyer

at al. (1985) for Cu in *P. leucopus* at a Zn smelter. This BCF value is near the low end of the range of BCFs measured for deer mice (BCF=0.03–0.27). Available tissue-soil BCF values for Zn in *A. sylvaticus* were 0.005 (Johnson et al. 1978) and 0.065 (Roberts and Johnson 1978), which are lower than the BCF values calculated for deer mice in this study (BCF=0.07–5.85).

Potential Effects at Higher Trophic Levels

The bioaccumulation data just presented indicate that trace element concentrations in tissues of selected grasses, invertebrates, and small mammals are elevated relative to reference conditions. However, elevated tissue concentrations of chemicals do not necessarily cause adverse effects in populations or communities (Warren-Hicks et al. 1989). To evaluate this issue further, bioaccumulation data are interpreted to characterize potential risks to selected receptors. For effects endpoints that were measured directly in deer mice, empirical relationships demonstrate that the observed concentrations of trace elements in abiotic and biotic media do not cause reductions in embryo implantation, litter size, or population catch (see ARCO 1994). For wildlife species, such as white-tailed deer and predatory birds, risk estimates based on food web exposure models are used to interpret data on trace element concentrations in soil and key food web species.

Food Web Exposure Models—In this section, the risks associated with the observed trace element concentrations in soil and biota are estimated from food web exposure models for selected receptors. The general model for estimating wildlife exposure to trace elements is described in the *Methods* section. Red fox, white-tailed deer, and American kestrel were selected to represent wildlife species from several taxa and various trophic levels. Each receptor was assumed to spend a portion of its time in trace elements-enriched areas and a portion of its time in areas with background concentrations of trace elements. For the latter areas, the mean concentrations of trace elements in soil for reference stations sampled during this study were used to represent background conditions. For trace elements-enriched areas, each receptor's time was also divided between various habitats (i.e., riparian, transitional, and agricultural).

Exposure to trace elements in soil and food was estimated for receptor species from trace element concentration data or BCFs obtained in this investigation, literature data on dietary requirements, and assumptions for selected exposure variables such as gastrointestinal absorption efficiency for trace elements. Because data on concentrations of trace elements in willow were available only for Silver Bow Creek stations and BCFs were not available to predict willow tissue concentrations based on soil concentrations of trace elements, exposure of white-tailed deer was evaluated only at Silver Bow Creek. Caloric requirements were modeled on the basis of body size, predicted metabolic rates, and assimilation efficiencies. Data for diet composition and body weight of each receptor were taken from the literature (see complete list of references in ARCO 1994). The primary components of the modeled diets were small mammals for red fox, grasses and willows for white-tailed deer, and insects and small mammals for American kestrel. Although red fox may also consume some terrestrial invertebrates (≤5 percent of diet on annual basis [Korschgen, 1959; Knable, 1974]), this component of the diet is negligible based on the potential intake of trace elements from ingestion of small mammals and soil. Therefore, invertebrates were not included in the model diet of red fox. Soil ingestion was estimated as 2.0 percent and 2.8 percent of dietary intake for red fox and white-tailed deer, respectively, based on studies by Beyer et al. (1994). Soil ingestion by American kestrel was assumed to be negligible because this species does not spend time in contact with the soil. The gastrointestinal absorption efficiency for each trace element was based on the assumption that the bioavailability from dietary sources in the wild is reduced relative to dietary exposures in the laboratory because trace elements may be relatively tightly bound to protein, cellulose, or other cellular substances in the wild in comparison with the matrix effects for the soluble form of a trace element spiked into laboratory food or water. Absorption of trace elements associated with tailings, smelter emissions, or other particulate mining-related waste is limited by the incorporation of the trace element into mineral matrices (e.g., Davis et al. 1992).

Evaluation of Model Results—Exposure estimates for the selected receptors were compared with literature-based TRVs using a hazard quotient to evaluate risk to the typical individual (Fig. 10). The results of the food web exposure models indicate that enrichment of trace elements in soils, plants, invertebrates, and

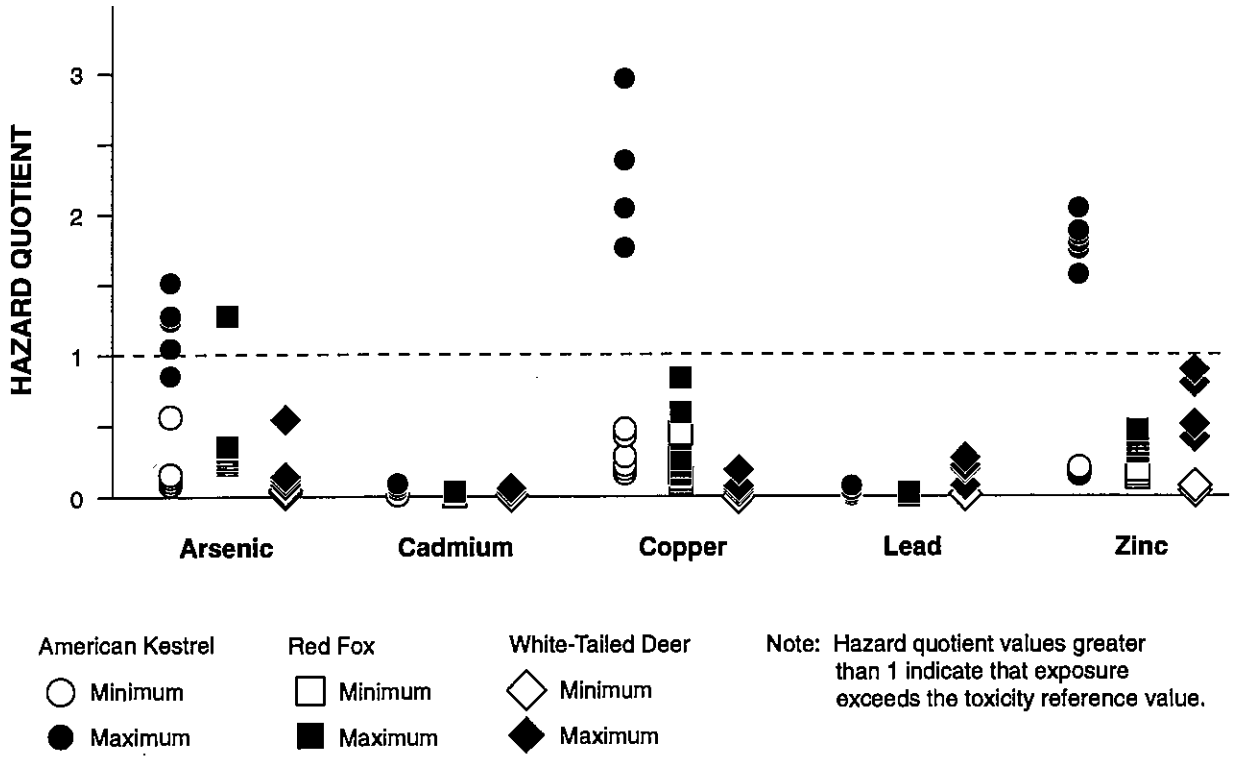

Figure 10. Hazard quotients for wildlife.

mice in the study area does not pose a significant risk to receptor species. However, comparisons of chemical intake estimates to TRVs derived from laboratory tests resulted in a few hazard quotients greater than 1.0. Although estimates of As and Cu intake by American kestrel exceeded the minimum TRVs, these hazard quotients were all less than 10, which was the uncertainty factor used to derive the TRV. Based on the magnitude of the exceedances, exposures to As and Cu are not expected to adversely affect local populations of American kestrel.

Uncertainty Analysis

Uncertainties in this screening evaluation of trace elements risk to wildlife of the UCFRB relate to the selection of receptors, the derivation of TRVs, and the estimation of exposure. The primary uncertainty associated with characterization of risks to individual receptors is associated with TRVs. Because of the uncertainties associated with TRVs, these values are probably underestimated (i.e., overprotective) by use of the uncertainty factors. Realistic assumptions and site-specific data on trace element concentrations in soil and biota were used in the exposure models (Table D-1, ARCO 1994). Nevertheless, uncertainty associated with exposure estimates could be approximately an order of magnitude. Despite the uncertainty associated with hazard quotients for individuals, the risk at the population level for all of the selected receptors is negligible. None of the hazard quotients for white-tailed deer and only one for red fox exceeded 1.0 (the exceedance for fox was minor). Hazard quotients for American kestrel exceeded 1.0 only in cases where a conservative uncertainty factor was used in the derivation of the TRV to account for interspecies extrapolation. Thus, the level of confidence is high for the conclusion that effects of trace elements on wildlife populations are not expected according to this investigation.

Conclusions

A major conclusion of this investigation is that the risk associated with existing levels of enrichment of trace elements in soils and terrestrial food webs of the Clark Fork Superfund Site is negligible for populations of

key ecological receptors, such as foxes, white-tailed deer, and predatory birds. Specific conclusions are summarized below:

- Risk to omnivorous small mammals such as the deer mouse is not significant. The relative abundance and reproductive condition of the deer mouse was normal, even in areas of high trace elements enrichment.
- Risk to raptor species (e.g., American kestrel) and other predators (e.g., red fox) that feed on small mammals such as the deer mouse is not significant.
- Risk to large herbivores (e.g., white-tailed deer) related to trace elements bioaccumulation in plant tissues is not significant.
- Risk to wildlife species that feed on terrestrial invertebrates is not significant based on the American kestrel food web exposure model.

References

ARCO. 1992. Clark Fork River Superfund site investigation quality assurance project plan. Prepared for ARCO, Anaconda, MT. PTI Environmental Services, Bellevue, WA.

ARCO. 1994. Regional ecorisk field investigation, Upper Clark Fork River Basin. Prepared for ARCO, Anaconda, MT. PTI Environmental Services, Bellevue, WA.

Beyer, W.N., O.H. Pattee, L. Sileo, D.J. Hoffman, and B.M. Mulhern. 1985. Metal contamination in wildlife living near two zinc smelters. Environ. Pollut. 38:63.
http://dx.doi.org/10.1016/0143-1471(85)90094-7

CAS. 1990. Sample preparation of biological tissue for heavy metal analysis by AA or ICP. Columbia Analytical Services, Inc., Kelso, WA.

Davis, A., M.V. Ruby, and P.D. Bergstrom. 1992. Bioavailability of arsenic and lead in soils from the Butte, Montana, mining district. Environ. Sci. Technol. 26(3)461–468.
http://dx.doi.org/10.1021/es00027a002

Johnson, H.E., and C.L. Schmidt. 1988. Clark Fork Basin Project status report and action plan. Clark Fork Basin Project, Office of the Governor, Helena, MT.

Johnson, M.S., R.D. Roberts, M. Hutton, and M.J. Inskip. 1978. Distribution of lead, zinc, and cadmium in small mammals from polluted environments. Oikos 30:153-159.
http://dx.doi.org/10.2307/3543536

Kabata-Pendias, A., and H. Pendias. 1992. Trace elements in soil and plants. Second Edition. CRC Press, Boca Raton, FL.

Knable, A.E. 1970. Seasonal trends in the utilization of major food groups by the red fox (*Vulpes fulva*) in Union County, Illinois. Trans. III. State Acad. Sci. 66:113–115.

Korschgen, L.J. 1959. Food habits of the red fox in Missouri. J. Wildl. Manage. 23:168–176.
http://dx.doi.org/10.2307/3797637

La Tier, A.J., P.I. Mulligan, R.A. Pastorok, and T.C. Ginn. In press. Bioaccumulation of trace elements and reproductive effects in deer mice (*Peromyscus maniculatus*). Submitted to American Society for Surface Mining and Reclamation for presentation at the ASSMR 12th Annual Meeting.
http://dx.doi.org/10.21000/JASMR95010003

Ma, W.C. 1994. Methodological principles of using small mammals for ecological hazard assessment of chemical soil pollution, with examples on cadmium and lead. pp. 357–371. In: Ecotoxicology of Soil Organisms. M.H. Donker, H. Eijsackers, and F. Heimbach (eds). Lewis Publishers, Boca Raton, FL.

Martin, M.H., and P.J. Coughtrey. 1976. Comparisons between the levels of lead, zinc and cadmium within a contaminated environment. Chemosphere 5:15–20 (not seen, as cited in Roberts and Johnson 1978). http://dx.doi.org/10.1016/0045-6535(76)90050-3

Martin, M.H., P.J. Coughtrey, and E.W. Young. 1976. Observations on availability of lead, zinc, cadmium and copper in woodland litter and the uptake of lead, zinc and cadmium by the woodlouse, *Oniscus asellus*. Chemosphere 5:313–318 (not seen, as cited in Roberts and Johnson 1978). http://dx.doi.org/10.1016/0045-6535(76)90005-9

McKinney, J., and R. Rogers. 1992. Metal bioavailability - EPA workshop identified research needs. Environ. Sci. and Technol. 26:1298-1299. https://doi.org/10.1021/es00031a603

Munshower, F.F. 1972. Cadmium compartmentation and cycling in a grassland ecosystem in the Deer Lodge Valley, Montana. Ph.D. dissertation, University of Montana. Missoula, MT.

Munshower, F.F. 1977. Cadmium accumulation in plants and animals of polluted and nonpolluted grasslands. Journal of Environmental Quality. V. 6:411-413. http://dx.doi.org/10.2134/jeq1977.00472425000600040016x

Pascoe, G. A., and R.J. Blanchet. 1993. Baseline ecological risk assessment, Milltown Reservoir Operable Unit, Milltown Reservoir Sediments Site. Prepared for U.S. Environmental Protection Agency Region 8. Environmental Toxicology International, Inc., Seattle, WA.

Pascoe, G.A., R.J. Blanchet, and G. Linder. 1994. Bioavailability of metals and arsenic to small mammals at a mining waste-contaminated wetland. Arch. Environ. Contam. Toxicol. 27:44-50. http://dx.doi.org/10.1007/BF00203886

Pastorok, R.A., M.K. Butcher, and T.C. Ginn. In press. Thresholds for potential effects of mining-related trace elements on riparian plant communities. Submitted to American Society for Surface Mining and Reclamation for presentation at the ASSMR 12th Annual Meeting. https://doi.org/10.21000/JASMR95010015

PTI. 1991. Evaluation of approaches for establishing root uptake factors. Final Report. Prepared for Washington Department of Ecology, Olympia, WA. PTI Environmental Services, Bellevue, WA.

PTI. 1993. Regional Ecorisk Field Investigation, Upper Clark Fork River Basin: Sampling and analysis plan. Prepared for ARCO, Anaconda, MT. PTI Environmental Services, Bellevue, WA.

Roberts, R.D., and M.S. Johnson. 1978. Dispersal of heavy metals from abandoned mine tailings and their transfer through terrestrial food chains. Environ. Pollut. 16:293–310. http://dx.doi.org/10.1016/0013-9327(78)90080-0

Roberts, R.D., and M.S. Johnson. 1979. Predator-prey relationships in the food chain transfer of heavy metals. pp. 104–114. In: Proc. of University of Missouri Annual Conference on Trace Substances in Environmental Health, Columbia, MO. D.D. Hempill (ed).

Roth, M. 1992. Metals in invertebrate animals of a forest ecosystem. pp. 299–328. In: Biogeochemistry of trace metals. D.C. Adriano (ed). Lewis Publishers, Boca Raton, FL.

Smith, G.J., and O.J. Rongstad. 1982. Heavy metals in mammals from two unmined copper-zinc deposits in Wisconsin. Bull. Environ. Contam. Toxicol. 27:28–33.

U.S. EPA. 1988. Contract Laboratory Program statement of work. Inorganic analysis, multi-media, multi-concentration, July 1988. SOW No. 788. U.S. Environmental Protection Agency, Environmental Monitoring and Support Laboratory, Las Vegas, NV.

Vallee, B.L. 1959. Biochemistry, physiology and pathology of zinc. Physiol. Rev. 39:443–490 (not seen, as cited in Roberts and Johnson 1978).

Van Hook, R.I. 1974. Cadmium, lead and zinc distributions between earthworms and soils: potentials for biological accumulation. Bull. Environ. Contam. Toxicol. 12:509–512.

Warren-Hicks, W., B.R. Parkhurst, and S.S. Baker. 1989. Ecological assessment of hazardous waste sites: a field and laboratory reference. EPA/600/3-89/013. Prepared by Kilkelly Environmental Associates, Raleigh, NC, and Western Aquatics, Inc., Laramie, WY.

AVIAN UTILIZATION OF SUBSIDENCE WETLANDS[1]

Jack R. Nawrot, Pat S. Conley, and Cody L. Smout[2]

Abstract: Diverse and productive wetlands have resulted from coal mining in the midwest. The trend from surface to underground mining has increased the potential for subsidence. Planned subsidence of longwall mining areas provides increased opportunities for wetland habitat establishment. Planned subsidence over a 180 meter (590 foot) deep longwall mine in southern Illinois during 1984 to 1986 produced three subsidence wetlands totaling 15 hectares (38 acres). The resulting palustrine emergent wetlands enhanced habitat diversity within the surrounding palustrine forested unsubsided area. Habitat assessments and evaluations of avian utilization of the subsidence wetlands were conducted during February 1990 through October 1991. Avian utilization was greatest within the subsided wetlands. Fifty-three bird species representing seven foraging guilds utilized the subsidence wetlands. Wading/fishing, dabbling waterfowl, and insectivorous avian guilds dominated the subsidence wetlands. The subsidence wetlands represented ideal habitat for wood ducks and great blue herons which utilized snags adjacent to and within the wetlands for nesting (19 great blue heron nests produced 25 young). Dense cover and a rich supply of macroinvertebrates provided excellent brood habitat for wood ducks, while herpetofauna and ichthyofauna provided abundant forage in shallow water zones for great blue herons and other wetland wading birds. The diversity of game and non-game avifauna utilizing the subsidence areas demonstrated the unique value of these wetlands. Preplanned subsidence wetlands can help mitigate loss of wetland habitats in the midwest.

Additional Key Words: subsidence, wetlands, wildlife utilization.

Introduction

Draining and clearing for agriculture has eliminated more than 85% of wetlands in the United States (Tiner 1984). Illinois has lost an estimated 90 percent of the original 3.2 million ha (8 million ac) of presettlement wetlands. Of the remaining 507,450 ha (1,253,891 ac) of Illinois wetlands, palustrine emergent and palustrine forested wetlands comprise 16 and 62 percent, respectively, of the total wetland acreage (Suloway and Hubbell 1994). Fortunately, the functions and values of wetlands are now appreciated, although the complex nature of these diverse and productive habitats are not completely understood. While current emphasis is placed on wetland protection and preservation, mitigation of some past wetland losses can be achieved through wetland development and enhancement on reclaimed mined lands. Opportunities for wetland/wildlife habitat enhancement on mined lands include shallow water wetlands and littoral zones associated with the reclamation of sediment ponds, final cuts and inclines, stream riparian zones, slurry ponds, within spoil depressions, and underground subsidence areas (Nawrot and Klimstra 1989).

As the emphasis shifts in the Illinois Coal Basin from surface to underground coal extraction methods the opportunities for subsidence associated wetlands will increase. Since commercial underground mining for coal began in 1810 more than 4,000 underground mines underlying more than 283,290 ha (700,000 ac) have operated in Illinois (Guither et al. 1985). Room and pillar underground mining has produced extensive areas of subsidence wetlands in southern Illinois. In one region five underground mines, operating from approximately 1910 to 1960, undermined more than 12,383 ha (30,600 ac) resulting in more than 364 ha (900 ac) of palustrine emergent and shrub-scrub wetlands (Baralt 1994). Although subsidence wetlands associated with room and pillar mines have been unplanned, they provide land use diversity and critical habitats for some of Illinois' threatened and endangered species (Sternburg and Nawrot 1994). As longwall

---

[1]Paper presented at the 1995 National Meeting of the American Society for Surface Mining and Reclamation, Gillette, Wyoming, June 5-8, 1995.

[2]Jack R. Nawrot, Associate Scientist, Pat S. Conley, Researcher II, and Cody L. Smout, Student Assistant, Cooperative Wildlife Research Laboratory, Southern Illinois University, Carbondale, IL 62901-6504.