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Evaluation of the Ecological Risks Associated with Land Application of Municipal Sewage Sludge

FINAL REPORT

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EXECUTIVE SUMMARY

This report provides final results for the EPA-funded project *Evaluation of Ecological Risks Associated with Land Application of Sewage Sludge*. The project was initiated to provide a detailed, ecosystem-specific evaluation of the risks that contaminants and nitrogen in municipal sewage sludge present to terrestrial ecological receptors. The risk assessment focuses on four ecosystems: 1) northwestern Douglas-fir forest, 2) southeastern loblolly pine plantation, 3) eastern deciduous forest, and 4) southwestern semiarid rangeland. Terrestrial features of the systems are emphasized in this assessment; risks to aquatic life and potential eutrophication of streams are not considered. Constituents of sewage sludge that are the subject of this report include: arsenic, cadmium, copper, lead, mercury, molybdenum, nickel, selenium, zinc, chlorinated dioxins, chlorinated dibenzofurans, PCBs, and nitrogen. The risk assessment is organized according to the EPA ecological risk assessment (ERA) framework. Under the ERA framework, an ecological risk assessment consists of four components: problem formulation, exposure assessment, effects assessment, and risk characterization. In addition, research needs are listed.

The primary exposure assumptions are:

- The concentration of contaminants in municipal sewage sludge is the 95% upper confidence limit about the mean contaminant concentration in the 1988 National Sewage Sludge Survey.
- Sewage sludge is applied at 40 Mg/ha in a single treatment.
- Chemicals in sewage sludge are assumed to be completely and uniformly incorporated into the top 15 cm of soil.

Wildlife

Individual foxes, shrews, American robins, and western meadow larks are not at risk from a single application of sewage sludge at the assumed rate in any of the four ecosystems, and individual deer are not at risk in the Douglas-fir forest, loblolly pine plantation, or semiarid rangeland. Because populations consist of individuals and individual-level risks are unlikely, population-level risks to these endpoint organisms are also considered unlikely. Limited bioaccumulation data suggests that white-tailed deer in the eastern deciduous forest may be at risk from copper and/or zinc.

In addition to potentially toxic contaminants, sewage sludge contains plant nutrients which have been shown to increase plant growth and alter plant community composition and structure. These effects of application of sewage sludge on plants can indirectly affect the wildlife communities present at sludge-treated sites. Due to the minimal extent of contaminant risks estimated or observed at sludge-amended sites and because wildlife responses to plant and habitat-mediated effects of sludge application have been observed, it is likely that the plant and habitat-mediated effects of sludge application will predominate.

Plant community

Sewage sludge may present a hazard to plant growth based on zinc toxicity in the three forests, but much evidence from short-term tests points to the opposite conclusion. Sewage sludge may present a hazard to plant growth and survival in the rangeland ecosystem, based on zinc, copper and/or cadmium toxicity, or nitrogen-catalyzed growth combined with drought stress, but some evidence suggests that these elements are not toxic at the application rate in this study.

Herbaceous community composition and biomass are likely to change with sludge application to forests, but whether the change constitutes a risk depends on the management goals for the forest or plantation. For example, if managed for timber or paper pulp, the understory may only be important in relation to nutrient processes and water balance, as they affect the trees. Similarly, some of the observed changes in the rangeland may be viewed as positive if the ecosystem is managed for grazing land (e.g., increased plant biomass, increased foliar nutrients, potential reduced density of noxious species). Modeling is useful in predicting the increased biomass and production, but this effect would not likely be deemed an adverse effect under any management scenario for the three ecosystems. If the predicted change in overstory composition in the deciduous forest is considered undesirable, then this endpoint would also be termed at risk.

Soil invertebrate community

Sewage sludge has the potential to alter the soil invertebrate community in all three ecosystems, as a result of added nutrients. Metal and organic contaminants in sewage sludge are probably not responsible for the soil invertebrate community changes. The effect may be considered a risk if it is deemed that any community shift constitutes a risk. However, there is not enough evidence to evaluate impacts on the invertebrate role in litter decomposition or the duration of the community change. An evaluation of risks to soil invertebrates in the rangeland is not presented, given the paucity of data on toxicity to non-earthworm invertebrates.

Microbial processes

The total biomass of microorganisms is likely to increase with sludge application. Litter decomposition in all ecosystems is may accelerate with sludge application, though an increased decomposition rate may be beneficial if an increase in plant biomass is associated with sludge application. The balance between nitrification and denitrification and nitrogen fixation by cyanobacteria and rhizobia has been showed to be altered and reduced, respectively, at application rates higher than that assumed in this study. Nitrification may be reduced by zinc in sewage sludge. Changes in the balance of processes in the nitrogen cycle are likely in the Douglas-fir forest and in the eastern deciduous forest and are more uncertain in the loblolly pine plantation and in the rangeland.

Uncertainty

Numerous sources of uncertainty exist in the risk characterization for contaminants. Exposure assumptions have been made out of necessity, because this is not a retrospective risk assessment for a single site. Sewage sludges vary in their composition, and the extent of mixing of constituents of sludge with surface soil in various ecosystems and conditions and under various application scenarios is unknown. Estimates of risks to abundance and production of populations are often highly uncertain, particularly where no field data are available as lines of evidence. The long-term bioavailability and uptake of chemicals in sewage sludge is not well understood. The estimation of bioaccumulation of contaminants is not specific to ecosystem type, soil characteristics, or species of organism and is therefore not very precise.

1. PROBLEM FORMULATION

1.1 REGULATORY AND PROJECT BACKGROUND

Approximately 12,750 sewage treatment works produce 5.3 million dry metric tons of municipal sewage sludge annually in the United States. In the last 10 to 20 years, land application of municipal sewage sludge has increased, with up to 30% of smaller communities in the United States currently applying sewage sludge to land (Bastian 1994). Improvements in the quality of sludge resulting from pretreatment, reductions in metals content in the past 10 years, and new regulatory requirements to protect human health (Bastian 1994) have contributed to the increased interest in land application of sludge. Rising costs of waste disposal have increased the viability of long-distance transport of de-watered sludge to disposal sites (e.g., New York sludge transported to MERCO Joint Venture in the Chihuahuan Desert in Texas) and the possibility of contracts for biosolids management operations (Bastian 1994). Industrial process sludge, such as that from the pulp and paper or poultry industries is also being increasingly applied to forest lands, particularly in the southeastern United States.

The Clean Water Act (CWA) of 1977 directed the EPA to establish standards for use and disposal of municipal sewage sludge. These standards, expressed as quantitative limits for pollutants in sludge, are intended to protect public health and the environment from any reasonably anticipated adverse effects. While criteria to protect human health have been established (EPA 1993c), less attention has been given to terrestrial ecosystems receiving the waste. The *Technical Support Document for Land Application of Sewage Sludge* (TSD, EPA 1992a) provides a generalized assessment of exposure and effects of contaminants in sludge on plants, soil organisms, vermivorous (worm-eating) wildlife, and herbivores. This basic ecological assessment does not address differences among ecosystems. An ecosystem-specific assessment of risks to biotic populations, communities, and ecosystem processes by contaminants and nutrients in sewage sludge has not previously been undertaken.

This project was initiated to provide a detailed, ecosystem-specific evaluation of the risks that contaminants and nitrogen in sewage sludge present to terrestrial ecological endpoints. An expert workshop was held at Oak Ridge, TN in April, 1995 to define the scope of this project. At the conclusion of this workshop, it was decided that this risk assessment should focus on four ecosystems: 1) northwestern Douglas-fir forest, 2) southeastern loblolly pine plantation, 3) eastern deciduous forest, and 4) southwestern semiarid rangeland. Terrestrial features of the systems are emphasized in this assessment; risks to aquatic life and potential eutrophication of streams are not considered. Comparisons of risks in sewage sludge-treated forests and rangeland and those in ecosystems managed for other purposes are not made. Sources of data for use in the assessment were identified and include field studies undertaken by the U. S. Forest Service Rocky Mountain Forest and Range Experiment Station, the University of Washington, Texas Tech University, Colorado State University, Savannah River Plant, and several other institutions. Issues of consensus from the expert workshop are available in the summary of the workshop (Barnthouse et al. 1995).

This risk assessment is organized according to the framework presented by EPA (1992c). Under this framework, an ecological risk assessment consists of four components: problem formulation, exposure assessment, effects assessment, and risk characterization.

In the problem formulation phase of an ecological risk assessment (this section, Sect 1), the goals, breadth, and focus of the assessment are established. The abiotic and biotic segments of the affected environment are described; the spatial extent of the problem is defined; stressors, such as heavy metals, organic chemicals, or nutrients, are identified; endpoints appropriate to evaluate ecological effects of the chemicals are selected; and conceptual models are developed that describe pathways by which nutrients and contaminants move through the abiotic and biotic environment, expose plants and animals, and induce effects.

In the exposure assessment, the transport and transformation of contaminants and other stressors and their contact with valued species, communities and ecosystem processes referred to as "assessment endpoint" entities are evaluated. Pathways by which plants and animals are exposed to contaminants and nutrients are identified and quantified.

The purpose of the effects assessment is to assess qualitatively and quantitatively the relationship between chemical exposure and effects on plants, animals, and ecosystems. These effects may be either direct or indirect and are evaluated using a combination of biological survey data, conventional toxicity data, ambient media toxicity test data, and bioindicators. Although data have been obtained from field sites where municipal sewage sludge was previously applied, this project has not included directed field studies in its scope.

The final step in performing an ecological risk assessment is risk characterization. In this phase, data from the exposure and effects assessments are combined to describe the risks to selected assessment endpoints. In addition, all assumptions, strengths, and weaknesses of various analyses and their associated uncertainties are summarized and discussed. Risk characterization combines information about exposure to contaminants and other stressors with information about effects of contaminants to quantitatively and qualitatively estimate risks. Risk characterization for ecological risk assessments is performed by weight of evidence (EPA 1992c). That is, rather than simply modeling risks, ecological risk assessors examine all available and appropriate data from chemical analyses, toxicity tests, biological surveys, and bioindicators to estimate the likelihood that significant effects are occurring or will occur and to describe the nature, magnitude, and extent of effects on the designated assessment endpoint entities (Sect. 1.4). Section 1.4 describes a weight-of-evidence approach for estimating risks based on combining individual lines of evidence.

This chapter is the problem formulation exercise for the assessment of ecological risks associated with the land application of sewage sludge. The report is organized into the following components:

1) Sources and Stressors: Data and statistics about particular contaminants in sludge from the 1988 National Sewage Sludge Survey.
2) Baseline Ecosystem Descriptions: Information about vegetation, wildlife, geography, soils, nutrient cycles, and management practices
3) Conceptual Models: Depiction of working hypotheses of how contaminants and/or nutrients in sludge may be transferred, transformed and/or cause effects.
4) Assessment Approach: Description of how the information will be obtained and used during the assessment, including sources of data and models.

1.2 SOURCES AND STRESSORS

Municipal sewage sludges contain a great number of organic and inorganic constituents. At sufficiently high concentrations, some of these chemicals may be toxic to exposed biota. In addition, municipal sewage sludges contain significant levels of nutrients such as nitrogen that may cause significant changes in terrestrial ecosystems, some of which may be viewed as hazards.

Results of the 1988 National Sewage Sludge Survey (NSSS; EPA 1990) were used as representative concentrations of chemicals in municipal sewage sludge. This survey reports the concentrations of 419 analytes (e.g., volatile and semivolatile organic chemicals, pesticides, PCBs, dioxins, dibenzofurans, metals, and other inorganic chemicals) from over 200 Publicly Owned Treatment Works (POTWs) from throughout the United States (EPA 1990, EPA 1992c).

At the April, 1995 project workshop held in Oak Ridge, TN (Sect. 1.1), 12 chemicals of potential concern in municipal sewage sludge were identified. These include: arsenic, cadmium, copper, lead, mercury, molybdenum, nickel, selenium, zinc, chlorinated dioxins, chlorinated dibenzofurans, and PCBs. Additional chemicals, such as nitrogen (as nitrate, nitrite, and total Kjeldahl N), phosphorus, potassium, aluminum, calcium, iron, and magnesium were added to the list of assessment chemicals to facilitate the evaluation of nutrient effects.

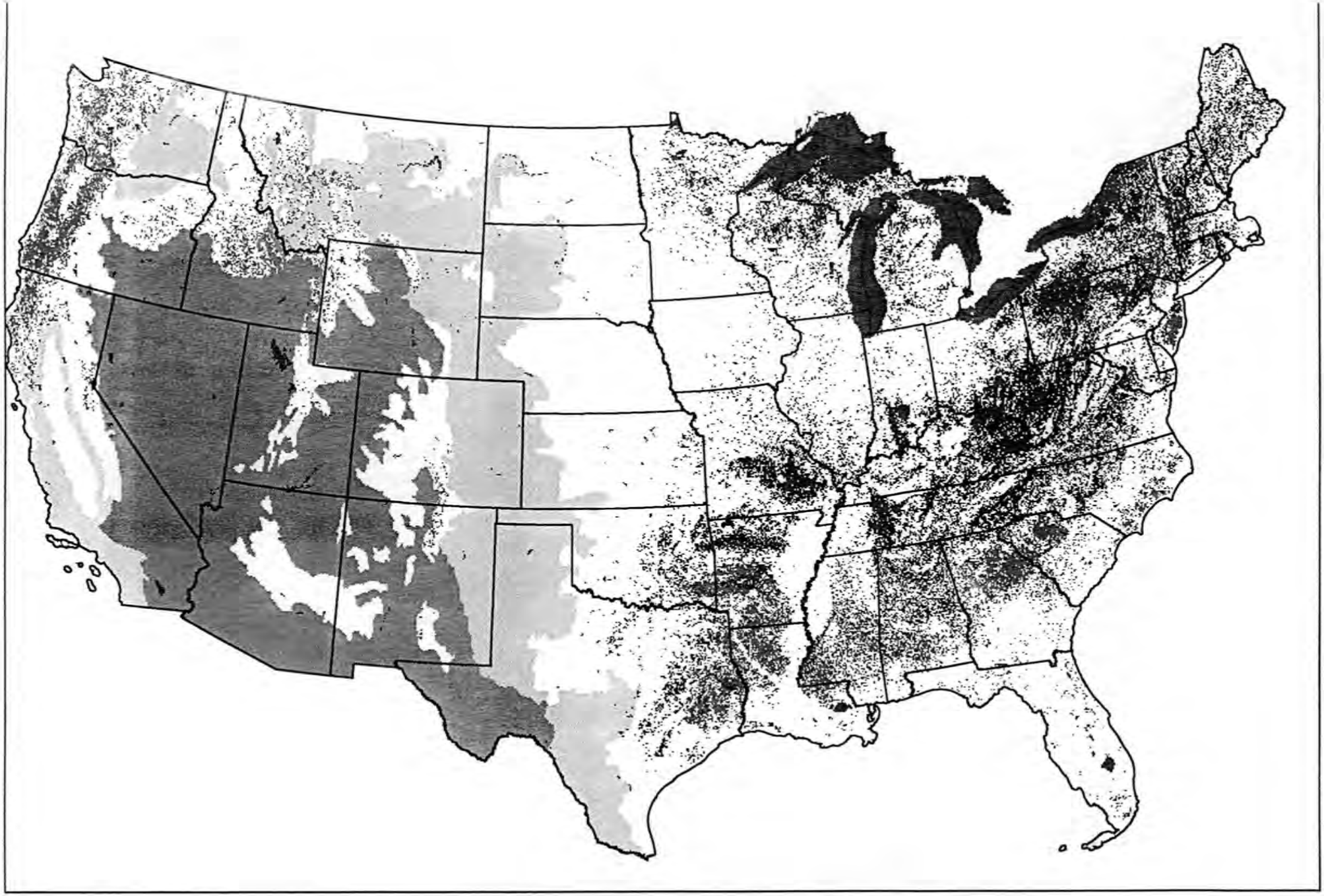
Summary statistics for the chemicals of potential concern in sludge are presented in Sect. 4.1. All data were assumed to be lognormally distributed (EPA 1992c), except for nitrogen, which was observed to be normally distributed. Samples of sludge that received only primary treatment were excluded from statistical calculations. Samples from 177 POTWs were used. Concentrations of chemicals from multiple samples at a POTW were averaged, regardless of treatment regime. All treatment plants were treated equally in statistical manipulations, regardless of the treatment flow rate. Upper 95% and lower 95% confidence limits (UCLs and LCLs) around the mean were calculated for each chemical using a Maximum Likelihood Estimator (MLE) approach to account for non-detected chemicals (EPA 1992c). UCLs represent a conservative upper bound for the mean concentration of chemicals in sludge and are used to screen for potential effects of the chemical. For highly skewed data, the UCL is generally biased and may exceed the maximum observed value. In these situations, the maximum observed value is used for the initial screening for potential toxic effects. The maximum observed value in the NSSS was greater than the 95% UCL for all chemicals considered, except PCB-1254, where both the mean and the 95% UCL exceeded the maximum observed value.

The water content of municipal sewage sludge may vary from 1% to 99% (EPA 1992d). Therefore, to facilitate comparison between POTWs, the NSSS assumed all that chemicals are associated exclusively with solids in sludge. Analyses reported in EPA (1992d) suggest this assumption to be valid. Therefore all chemical concentrations in sludge were standardized to a dry weight basis.

Measurements of the pH of municipal sewage sludge were not included in the NSSS.

1.3 ENVIRONMENTAL DESCRIPTIONS

As stated in Section 1.1, this assessment focuses on four ecosystems: northwestern Douglas-fir forest, southeastern loblolly pine plantations, eastern deciduous forest, and semiarid rangeland. The geographic scope of the assessment is roughly depicted on Map 1. The incidence of Douglas-fir forest, loblolly-shortleaf pine, oak-hickory and maple-beech-birch



- Other
- Loblolly-shortleaf pine
- Arid Western Xeric
- Douglas-fir
- Oak-Hickory/Maple-Beech-Birch
- Semi-Arid Western Xeric

forests are taken from the U. S. Forest Service 1993 RPA data. The boundaries of the loblolly pine alone are probably somewhat smaller than the area depicted. The arid western xeric and semi-arid western xeric ecoregions are taken from *Omernick's Aggregated Ecoregions* and certainly represent more than the "southwestern" semiarid rangeland in this assessment but may represent the region to which that assessment is applicable.

Brief summaries of the geography, climate, soils, biota, and management practices employed in each ecosystem are presented below. Mean monthly precipitation and temperature values for the three forests are presented in Sect. 2.2. It is assumed that the types of ecosystem processes (if not the relative importance) are comparable among the four ecosystems for which impacts of sludge application are being assessed. Thus, the conceptual models presented in Section 1.4 represent all four ecosystems.

1.3.1 Northwestern Douglas-Fir Forest

1.3.1.1. Geography, climate, and soil classification

Douglas-fir forest is found near the Northwestern Pacific Coast of North America, extending from Northern California to Southern Alaska. Although the Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) is found in most forested regions of this strip, the trees are dominant in well-drained areas at low and middle elevations (Shumway 1981). Most empirical data on land application of municipal sewage sludge in this ecosystem type is from Pack Forest, managed by the University of Washington. This research forest is located about 120 km south of Seattle, WA, in the foothills of the Cascade Mountains near Eatonville, at an elevation of 250 to 260 m (Gessel et al. 1990). For the purpose of this risk assessment, all information obtained from the Pack Forest is assumed to be representative of other Douglas-fir-dominated forest areas of the Pacific Northwest.

The climate of the coastal Douglas-fir forest has been described as a "winter wet-summer dry" maritime climate (Shumway 1981). Pack Forest has an annual average maximum temperature of 15.6° C and average minimum temperature of 5.8° C. Precipitation averages 100 mm per month, 680 mm (total) from October through March and 320 mm (total) from April through September (Gessel et al. 1990). In the Douglas-fir region in the state of Washington, only 4% of the precipitation falls in July and August. The evapotranspiration potential, which is associated with temperature, is highest during the summer, when precipitation is lowest (Shumway 1981). Potential losses of nutrients by leaching from Douglas-fir forests are reduced by evapotranspiration (Fredriksen and Harr 1981). Annually, 10-15% of the precipitation in the Douglas-fir forests is generally intercepted by plant surfaces and evaporated (Fredriksen and Harr 1981).

The soils at the Pack Forest are approximately 15 m deep, in the Everett series of the Inceptisol Order, characterized as a gravely loamy sand, and fully described, including pH, cation-exchange capacity, and quantities of nitrogen fractions, in Gessel et al. (1990). Because of steep slopes and young geology (Fredriksen and Harr 1981), the Pacific Northwest generally has high natural rates of erosion. However, rates of erosion are not necessarily greater than those of soil formation (Janda 1981).

1.3.1.2 Nutrients

Productivity of Douglas-fir forests is usually considered to be nitrogen-limited. Evidence includes the consistent responses to fertilizers and the correlation of various measures of soil nitrogen with the site index for Douglas-fir (Heilman 1981). (Site index is a term that expresses the height of dominant trees projected to a particular stand age (Pritchett 1979).) Miller et al. (1986) claim that tree growth at 70% of Douglas-fir sites is limited by soil nitrogen supply. The C:N ratio in the A1 (incorporated humus) horizon of soils in the typical Douglas-fir region is generally between 15 to 1 and 25 to 1 (Youngberg 1981). Tables and references for nutrient content of northwest tree species litterfall and components of litterfall can be found in Youngberg (1981).

Gessel et al. (1990) examined the role of soil water in controlling growth of Douglas-fir and determined that nutrients were far more important, even in the shallow soils on south-facing slopes at Pack Forest. Precipitation has a weak relationship with site index in the Douglas-fir region (Steinbrenner 1981). Steinbrenner (1981) describes factors that account for variation in site index for Douglas-fir. Of the factors examined, the depth of the A soil horizon (which reflects incorporated organic matter) accounts for the greatest percentage of variation.

Marshall (1991) speculates that the some soils of the Pacific Northwest may exhibit deficiencies in phosphorus, potassium, sulfur, boron, copper, iron, and/or zinc. Calcium and magnesium may also be low, though responses to fertilizers (at least prior to 1981) have not been shown to occur (Heilman 1981). Deficiencies of boron, sulfur, and phosphorus have been observed, following treatment with nitrogen, and references are available in Weetman et al. (1992). Tables of annual leaching losses and associated references can be found in Cole and Johnson (1981).

1.3.1.3 Soil biota and processes

Decomposition proceeds rapidly in the Douglas-fir forest. As in other forests, wood and litter are broken down by fungi with bacteria, mites, insects, centipedes, millipedes, earthworms, and other invertebrates. Nitrogen generally limits the rates of the decomposition process (Trappe and Bollen 1981).

As in all soils, the forms of nitrogen available to plants are ammonium, nitrite, and nitrate. Ammonification is the conversion of organic nitrogen to ammonium, and nitrification is the further conversion to nitrite and nitrate. Nitrite is rapidly converted to nitrate in well-aerated soils. In forest soils, nitrification is generally slower than in agricultural soils (Heilman 1981) because of reduced soil temperatures, excessively high or low soil moisture, inadequate soil aeration, and a high C:N ratio. Among the nitrogen-fixing plants in the Douglas-fir forest are red alder, snowbrush ceanothus, and lichens, which can contribute substantial nitrogen to the system (Fredricksen and Harr 1981). Legumes typically are not shade-tolerant and are therefore not commonly found in well-stocked stands (Trappe and Bullen 1981).

Mycorrhizae, whose functions are to obtain sugars and carbohydrates from host plants and to absorb and translocate water and nutrients to the plants, are extremely important in the Douglas-fir region. Low mycorrhizal biomass associated with Douglas-fir often leads to phosphorus deficiency (Trappe and Strand 1969). Mycorrhizal fungi are found on most fine roots. Taxa include the *Ascomycetes* and *Basidiomycetes* on Douglas-fir and *Endogonaceae* (*Phycomycetes*) on ferns and maple (Trappe and Bollen 1981). Grasses and weed roots share mycorrhizae with the ferns and maple (Trappe and Bollen 1981). Squirrels, red-backed voles,

chipmunks and slugs eat the mycorrhizal mushrooms and truffles on Douglas-fir. The *Endogonaceae* are eaten more by earthworms, millipedes, deer mice, and jumping mice (Trappe and Bollen 1981). Whether or not root-feeding aphids and nematodes are attracted to tree roots or prevented from contacting the roots depends to a certain degree on the particular mycorrhizal species present (Trappe and Bollen 1981).

1.3.1.4 Vegetation

The northwestern Douglas-fir forest is segregated into ten vegetative zones based upon species composition (Franklin 1981). Douglas-fir is the dominant tree canopy species in 8 zones. Other canopy tree species include western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), western redcedar (*Thuja plicata* Donn), sitka spruce (*Picea sitchensis* (Bong.)Carr.), grand fir (*Abies grandis* (Dougl.) Lindl.), and white fir (*Abies concolor* (Gord. & Glend.) Lindl.) (Frankin 1981). Because Douglas-fir is shade intolerant (Franklin and Dyrness 1973), the understory in Douglas-fir forests are dominated by other species. These include red alder (*Alnus rubra* Bong.), salal (*Gaultheria shallon* Pursh), big-leaf maple (*Acer macrophyllum* Pursh), rhododendron (*Rhododendron* spp.), pacific madrone (*Arbutus menziesii* Pursh), and swordfern(*Polystichum munitum* (Kaulf.) Presl.). Detailed lists of typical species composition for the vegetative zones, by percent cover and frequency, are presented in Franklin and Dyrness (1973).

The Pack Forest is within the western hemlock zone of the Northwestern Douglas-fir forest. Forest cover at this site is a patchwork of various age forests resulting from both timber harvest and fires. Dominant age classes include 1-15-yr old clearcuts planted in Douglas-fir, 40-70-yr old second growth stands, and old growth stands >150-yrs old (Anderson 1985). Species composition of the second growth stands include pure Douglas-fir, mixed conifers (Douglas-fir, western hemlock, and western redcedar), and mixed conifer-hardwood that includes red alder.

1.3.1.5 Wildlife

Biological diversity within the Douglas-fir forest is high. At the H. J. Andrews Experimental Forest, a LTER site located in the Willamette National Forest in Oregon, 20 species of reptiles and amphibians, 53 mammalian species, and 88 avian species have been identified (OSU 1996). Common bird species in second-growth Douglas-fir stands include rufous hummingbird (*Selasphorus rufus*), red-tailed hawk (*Buteo jamaicensis),* ruffed grouse (*Bonasa umbellus*), dark-eyed junco (*Junco hyemalis*), and American robin (*Turdus migratorius*). Common mammal species include Trowbridge shrew (*Sorex trowbridgii*), deer mouse (*Peromyscus maniculatus*), bobcat (*Lynx rufus*), and black-tailed deer (*Odocoileus hemionus columbianus*) (OSU 1996).

1.3.1.6 Management practices

Douglas-fir forests are generally managed for timber production. Activities may include: clearcutting, thinning, nitrogen fertilization, pesticide use, and, at the Pack Forest, the construction of haul roads to facilitate sludge application. In addition, nutrient relations within trees and soils are dependent on the age of the stand.

Douglas-fir forests are commonly fertilized with nitrogen. Among nutrient additions, only nitrogen has given consistent responses, in terms of increased productivity (Heilman 1981). As of 1981, granular urea (46 percent N) was the most common source, with 168 or 224 kg N/ha

being the most common application rates (Moore and Norris 1981). With two applications of urea nitrogen, a site III Douglas-fir stand produces about 90 m^3 in supplementary timber volume during a 50-year rotation (McMahon 1992). The factors that influence growth response to fertilizer, including treatment, stand characteristics, and site characteristics, are summarized by Gessel and Atkinson (1981). Treatment refers to the chemical product, as well as the form, amount, and weather conditions before and after treatment. Stand characteristics include species present, stocking, age of trees, growth rate, and size class distribution in the stand (Gessel and Atkinson 1981). Site characteristics discussed above include nutrients and moisture in soil, precipitation, elevation, and aspect.

Harvesting timber has substantial effects on the nutrient balance in the Douglas-fir forest: 1) uptake of nutrients by trees and other plants is slowed, 2) the decomposition is rapid, because of increased heat and moisture at the forest floor, 3) soil erosion may increase, and 4) increased leaching of soluble nutrients may occur (Fredriksen and Harr 1981). During log removal, upper soil is disturbed, fresh branches and Douglas-fir needles are added to soil, and the soil surface is exposed to sunlight (Trappe and Bollen 1981). Mycorrhizal fungi dependent on tree roots lose their food source and die. Various shrubs and herbs may remain on the site, or, if clearcut and burned, herb species such as fireweed will establish (Fredriksen and Harr 1981). After burning, cation leaching (e.g., Ca, Mg, K, Na) may increase two to three-fold (Fredriksen and Harr 1981), which may have more implications for exposure to toxic metals in sludge than to forest productivity. pH may increase slightly (Trappe and Bollen 1981). Burning of soil organic matter can also substantially increase erosion because of the breakup of soil aggregates (Youngberg 1981). Alders may dominate a burned site for several decades before Douglas-firs take over.

Thinning of dense stands can sometimes optimize moisture and temperature conditions for litter decomposition, humus formation, and the release of nutrients for uptake (Youngberg 1981).

Chemicals used in forestry are described in Moore and Norris (1981). These primarily include fertilizers, pesticides (primarily herbicides used to suppress competing species), and fire retardants.

1.3.2 Southeastern Loblolly Pine Plantation

1.3.2.1 Geography, climate, and soil classification

The southern pine region of the southeastern United States extends from Virginia to Florida and west to Texas and Oklahoma, defined mostly by 17 million hectares of natural and planted loblolly and slash pine (Allen et al. 1990). Loblolly pine (*Pinus taeda* L.), the focus of this assessment, is largely planted in the Piedmont and Coastal Plain physiographic regions (Allen et al. 1990). Because the Piedmont is where most of the plantations are most productive, Athens, GA has been selected as the location in which to focus the risk assessment. Most data on land application of municipal sewage sludge in loblolly pine plantations is from the Savannah River Plant, located on the Upper Coastal Plain in southwestern South Carolina. The land has been forested since 1953, when it was converted from agriculture (Davis 1989). Experimental data on slash pine plantations are available from a University of Florida sludge application study site in northwest Florida near Jay (Riekerk and Lutrick 1986).

The climate of the southeastern pine region is generally considered humid, with annual rainfall of 1000 to 1650 mm (Allen et al. 1990). According to measurements taken in the U. S. Historical Climatology Network (NOAA), Athens,GA has a mean annual precipitation of 1330 mm. The Savannah River Plant has a mean annual precipitation of 1020 mm (Langley and Marter 1973).

The Cecil soil series in the Piedmont region is in the clayey, kaolinitic, thermic family of the Hapludults (Bruce et al. 1983). Cecil soils are found in Athens, GA, the focus of the loblolly assessment, and extend from Georgia to Alabama, South Carolina, and Virginia. Ap horizons vary in clay content from 6% to 38%, leading to large variations in soil water retention and transport characteristics. Tables of soil water, bulk density, and particle size may be found in Bruce et al. (1983).

Soils of the Coastal Plain of the southeastern United States, another loblolly region, are generally very sandy with poor cation exchange and water holding capacities (Ridgeway et al. 1986). The soils of the Savannah River Plant are described in Davis (1989). Most of the study plantations are established in the Orangeburg loamy sand series. The soils are well to slightly excessively drained with between 200 and 500 mg/kg N in the top 10 cm of soil. The pH range is 5 to 5.5, and the cation exchange capacity is 0.01 to 0.02 mol/kg through the soil profile (Davis 1989). There is little erosion or surface runoff in pine plantations (Jorgensen et al. 1975). Allen et al. (1990) provide several references that suggest that soil water often limits pine growth. Recent research has shown that N is more limiting than soil water for loblolly pine growth on sandy Coastal Plain soils (Allen, personal communication).

1.3.2.2 Nutrients

As is true for all forests, natural sources of nutrients to the loblolly pine system include: 1) transfers within biomass; 2) products of the decomposition of organic matter; 3) wet and dry deposition; 4) nutrients in the stand canopy and on stems that are washed below; 5) nutrients obtained from soil minerals; and 6) occasional transfers from flooding (Switzer and Nelson 1972, Jorgenesen et al. 1975).

Nitrogen is the element most often limiting to forest production in the southeastern United States (Pritchett and Smith 1975). This element has been shown to be the immediate limiting nutrient in pine stands at the Savannah River Plant, based on responses to fertilization and the low concentrations of nitrogen in needles (Birk and Vitousek 1986). Waide and Swank (1975) demonstrate that loblolly pine plantations have smaller storage pools of nitrogen (abiotic and biotic) than the nearby Coweeta, NC oak-hickory forest. According to Allen et al. (1990), nitrogen limitation begins to be observed at crown closure. Fertilization with phosphorus or potassium may occasionally stimulate growth of pine. Phosphorus has been shown to be deficient on many Coastal Plain soils, with large increases in growth following fertilization (Allen et al. 1990). On the other hand, fertilization of loblolly with phosphorus often has no effect on growth (Jorgensen et al. 1975). Coastal Plain soils with deep, sandy profiles occasionally respond to potassium fertilization (Allen et al. 1990). When potassium does not increase growth, it has still been shown to increase foliar potassium in loblolly (Wells 1970).

Nutrient cycling in a pine plantation is closely linked to the stand age. The maximum uptake of nutrients in loblolly pine plantations occurs when the trees are between 10 and 15 years old, around the time of stand closure (Jorgensen et al. 1975). At this time, litterfall increases,

and the rate of nutrient uptake decreases (Wells and Jorgensen 1975). Nutrient accumulation in the forest floor is highest a few years after stand closure, with steady state occurring around 30 years (Jorgensen 1975). Switzer and Nelson (1972) provide estimates of the fraction of several nutrients that are cycling during the 20th year of a loblolly plantation. The foliage is the main agent in the nutrient recycling process, with the following order of nutrient requirements in loblolly: foliage >> stems > branches (Switzer and Nelson 1972).

Soils of the Southeast (Holechek et al. 1989) are the most prone to leaching in the United States. The potential for leaching in loblolly pine plantations is dependent on management practices. For example, leaching losses are not believed to be significant in an uncut pine plantation (Jorgensen et al. 1975), though leaching of nitrate has been observed following fertilization (Wells et al. 1975).

1.3.2.3 Soil biota and processes

Other than in the decomposition of litter, soil microbial processes are rather unimportant in the tight nutrient cycle of the loblolly pine plantations. Soils have low fertility and annual root growth is high (Waide and Swank 1975). Switzer and Nelson (1972) provide estimates of the contribution of various sources (e.g., litter decomposition, precipitation, soil mineralization) to nutrient requirements of loblolly pine, and the contribution of soil mineral and organic sources of nutrients is low. In the Piedmont, soil acidity, low organic matter in soil, and few legumes probably reduce nitrogen fixation to less than 1 kg/ha/year (Jorgensen et al. 1975). Also, nitrification is somewhat low in soils in the southeastern United States because of the low pH (Davis 1989). Soil nitrogen becomes available slowly, at a rate of 3 or 4 % annually (Jorgensen et al. 1975).

As in the Douglas-fir forest, wood and litter are broken down by fungi with bacteria, mites, insects, centipedes, millipedes, worms, and other invertebrates. Sandy soils and acidic soils, such as those found in the pine plantations, are not favorable habitats for earthworms (Pritchett 1979); thus, fewer earthworms would be expected than in the Douglas-fir region.

1.3.2.4 Vegetation

Loblolly pine (*Pinus taeda* L.) is the most significant species for timber and pulpwood production in the southeastern United States (Wells and Jorgensen 1975), with over 12 million ha in natural and planted stands (Allen et al. 1990). Other pines in the southeastern U. S. region include: slash (*P.* elliotti Engelm.), shortleaf (*P. echinata* Mill.), longleaf (*P. palustris* Mill.), pond (*P. serotina* Michx.), sand (*P. clausa* (Chapm.) Vasey), and Virginia (*P. virginiana* Mill.) pine (Allen et al. 1990).

Southern pines are pioneer species, with early growth of up to $35 m^3 ha^{-1} year^{-1}$ (Allen et al. 1990). The stability (resiliency and resistance) of pine plantations is low compared to an oak-hickory system (Waide and Nelson 1975). The volume of production in southern pine forests may range from 7 to 15 $m^3 ha^{-1}$ during a 25 to 30-year rotation. The site index of southern pine, defined by the dominant and co-dominant tree height at 25 years, is from 15 to 25 m (Allen et al. 1990).

At the Savannah River Plant plantations, the major part of the understory vegetation is herbaceous, with grasses secondary. Shrubvine and woody components are not so important

except in the presence of some herbicide treatments (Davis 1989). The total understory biomass decreases with stand age.

1.3.2.5 Wildlife

Because southern pine forests are considered to be subclimax, progessing to a climax pine-hardwood or hardwood forest type (Blair and Brunett 1976), there is considerable overlap between the wildlife communities in southern pine forests and those in eastern deciduous forests (see Sect. 3.3.5.). Many of the same species are common in both forest types. These include wild turkey (*Meleagris gallopavo*) (Palmer et al. 1993) and white-tailed deer (*Odocoileus virginianus*) (Pearson and Sternitke 1976).

Johnston and Odum (1956) censused breeding birds in various successional habitats (including loblolly/shortleaf pine forests) in the Georgia piedmont. They observed lower densities in pine forests than in preceding (shrub) or following (oak-hickory) successional stages. Common birds in the pine stands included northern cardinal (*Cardinalis cardinalis*), rufous-sided towhee (*Pipilo erythrophthalmus*), carolina wren (*Thryothorus ludovicianus*), and pine warbler (*Dendroica pinus*).

1.3.2.6 Management practices

Loblolly pine plantations are generally managed for timber production. Timber management activities may include: clearcutting, thinning, fertilization, and pesticide use. The method of removal of timber, ranging from debarked pulpwood to the entire tree with large roots, will have a tremendous effect on the nutrient balance at the site (Jorgensen et al. 1975). Davis (1989) provides references for the intensive forest management practices, such as whole tree harvesting and pine tree removal, that drain the nutrients from pine plantations.

Effects of fertilization on loblolly are discussed in section 3.2.2. In general, loblolly pine plantations are commonly fertilized with nitrogen and/or phosphorus, but there has been little response to potassium fertilization (Wells 1970). Over 403,000 ha of loblolly have been treated with phosphorus through 1987, though the treatment area has been almost exclusively industry land (Allen et al. 1990). As of 1990, thirty-four percent of the loblolly pine land was owned by the forest industry, fifty-eight percent by other private owners, and eight percent by public agencies (Allen et al. 1990).

Herbicide use is common in southeastern pine plantations for weed and hardwood control. Allen et al. (1990) estimate that herbaceous weed control and brush control (chemical site-preparation) are used on about 25,000 ha and 120,000 ha of pine plantation land, respectively. At the Savannah River Plant, some of the sludge-treated loblolly plots were previously treated with herbicide (Davis 1989). Pines typically show growth response to weed control associated with increased nutrient uptake, growth and reduced water stress (Carter et al. 1984, Allen et al. 1990). Competition with hardwood sprouts is harsher in establishment and 3-year old plantations than in older ones (Davis 1989). Prescribed burning in pine stands is generally carried out every 3 to 7 years.

Water levels are often managed in pine plantations to increase productivity and to aid in harvesting, regeneration, and fire control (Allen et al. 1990). As of 1990, over a million ha of plantation lands were drained (Allen et al. 1990).

Allen et al. (1990) provides references for management practices associated with regeneration. For example, bedding is applied on over 90% of poorly drained soils used by the forest industry (Allen et al. 1990).

1.3.3 Eastern Deciduous Forest

1.3.3.1 Geography, climate, and soil classification

The eastern deciduous forest is divided into four regions, as described in section 1.3.3.4: northern hardwoods, central hardwoods, southern oak-pine, and bottomland hardwoods. The boundaries of the ecosystem are the northern coniferous forest of southern Canada; the Atlantic coast; the Gulf coast and tropical forest of southern Florida; and the grassland, prairie and agriculture of the midwest and west (Yahner 1995). The assessment of the ecological effects of sludge application to eastern deciduous forest system will focus on the Hubbard Brook Experimental Forest (HBEF) in the White Mountains in central New Hampshire, part of the northern hardwoods region. Municipal sewage sludge was experimentally applied to part of that forest in 1975 and 1976 (Koterba et al. 1979), and long-term effects on the forest from nitrogen in sludge have been simulated previously (Crohn 1995b).

Bormann and Likens (1979) provide references for the classification of the HBEF as part of the Hemlock-White Pine Northern Hardwood Region between Nova Scotia and northwestern Minnesota and as redspruce-hardwoods in the New England section of the Northern Appalachian Division. Thus, Bormann and Likens (1979) argue that HBEF is part of an extensive forest type.

Most soils at HBEF are well-drained spodosols (haplorthods) of sandy loam texture, with a deep organic layer (3-15 cm) (Likens et al. 1977), and a pH of about 4.5. There is very little overland flow in this ecosystem (Likens et al. 1977).

1.3.3.2 Nutrients

The broad geographic range of eastern deciduous forests and the associated spatial variability in climate, soils and dominant vegetation suggests that the nutrient relations of this ecosystem type are not as easily generalizable as those of the loblolly pine plantation, Douglas-fir forest or semi-arid rangeland systems.

Most northern forests are nitrogen-limited (Aber et al. 1989). Evidence for nitrogen limitation includes the response of trees to only the nitrogen component of fertilizers (Aber et al 1989, Lea et al. 1980). Although the rule applies to most eastern deciduous forests, exceptions exist. For example, the Long-term Ecological Research (LTER) site at Coweeta, NC may be phosphorus-limited (Monk and Day 1988).

In addition, forests with a continual input of nitrogen from acid rain may cease to be nitrogen-limited. Aber et al. (1989) describe a phenomenon called "nitrogen saturation," in which atmospheric deposition in northern forests may cause nitrogen availability in nitrogen-limited systems to exceed plant and microbial uptake. Water or phosphorus may then limit net primary production. Aber et al. (1989) suggest that chronic additions of nutrients through deposition may produce greater changes in structure and function of forests than effects observed in pulsed fertilization studies.

Calcium uptake may be higher than that of nitrogen for most hardwoods (Pritchett 1979), but uptake varies by species. For example, flowering dogwood and tulip poplar concentrate calcium in leaves, while American beech and eastern hemlock are low in the element (Yahner 1995). Likens et al. (1996) have demonstrated a loss of calcium in soil at Hubbard Brook since 1970, which may be attributable to acid rain. The decline in annual forest biomass accumulation at Hubbard Brook since 1987 may be associated with a limitation of available calcium (Likens et al. 1996).

There is typically a net retention of nitrogen and phosphorus, a net loss of Ca^{2+}, Mg^{2+} and K^{+}, and either retention or loss of sulfur in deciduous forests (Johnson and Henderson 1989). Likens et al. (1977) provide a table of precipitation inputs and stream outputs of various nutrients in HBEF. As of 1977, large quantities of sulfur (6.1 kg/ha-yr) were entering the HBEF from the air (Likens et al. 1977).

Available light is a factor controlling productivity of trees and herbaceous plants in deciduous forests, though it is likely not limiting except on nutrient-rich, well-watered sites (Aber et al. 1989). Climax forests may have closed canopies in the spring and summer with occasional gaps where trees have died (Hicks and Chabot 1985). In typical deciduous forests, 50-80% of solar radiation reaches the forest floor in winter, in contrast to the 1-5% that reaches the litter in summer (Hicks and Chabot 1985). The end of the winter dormant period for woody and herbaceous species is signaled by temperature, and possibly by precipitation for the latter species (Hicks and Chabot 1985).

Pritchett (1979) reviews reports that demonstrate soil properties related to site productivity for various types of stands. For eastern hardwoods, depth to pan or mottling, surface and subsoil texture, nutrient and organic content, soil drainage, depth of A horizon and depth to water table are frequently related to productivity. The average annual net primary production at Hubbard Brook between 1956 and 1965 was 875 g of dry matter/m^2-yr aboveground and 1044 g/m^2-yr total (Bormann and Likens 1979).

1.3.3.3 Soil biota and processes

As in other forests, wood and litter are broken down by fungi with bacteria, mites, insects, centipedes, millipedes, earthworms, and other invertebrates. Litter decomposition has been shown to be nitrogen limited in northern deciduous forests (Melillo et al. 1984). Decomposition of the forest floor in cooler environments may be slow, with a thick organic layer, low pH, high fungal populations, few earthworms, and a high concentration of phenolic substances in litter (Waring and Schlesinger 1985). Deciduous forests in warm temperate climates typically have floors with rapid decomposition of nutrient-rich litterfall by bacteria, abundant earthworms and mixing of soil horizons (Waring and Schlesinger 1985). Also, forests with well-drained soils have rapid rates of decomposition and thin layers of leaf litter, compared to those with poorly drained soils (Yahner 1995). Leaf litter is important to shrews, rufous-sided towhees, and other wildlife that obtain soil invertebrates from the forest floor (Yahner 1995).

Nitrogen enters temperate deciduous forests primarily through symbiotic associations of early-successional tree roots and *Frankia* or *Rhizobium*, free-living bacteria in soil or litter, or lichens on the surfaces of trees and decaying wood (Waide et al. 1988). Likens et al. (1977) have suggested that nitrogen fixation may provide significant bioavailable nitrogen to the Hubbard Brook ecosystem, especially in dead wood. A 55-year-old ecosystem at Hubbard Brook has been

observed to obtain 70% of its nitrogen from nitrogen fixation (Bormann and Likens 1979). In Coweeta, NC, a southern deciduous forest, biological nitrogen fixation accounted for 75% of total nitrogen input (Todd, Waide, and Cornaby 1975). In the southern Appalachians, black locust is an early successional species whose association with *Rhizobium* can contribute significant quantities of fixed nitrogen (Waide et al. 1988). Alder is also an important nitrogen-fixing species (Yahner 1995).

The descriptions of ammonification, nitrification, and denitrification in Section 3.1.3 apply equally to the eastern deciduous forest. Nitrification proceeds much more rapidly in well-mixed forest floors with mineral soils than in poorly mixed material (Aber et al. 1989). Nitrification does not occur in soil horizons with pH below 4.2 or in the presence of an O horizon (Aber et al. 1989). Despite a strong potential for nitrification, only about 10 to 20% of ammonium in soils of an HBEF stand ($\geq$ 15 yr post-clearcut) are nitrified to nitrate (Melillo 1977). Microbial denitrification may be an important transformation process in Hubbard Brook (Likens et al. 1977). Potential denitrification has been shown to be 200 times the magnitude of stream water losses at Coweeta, NC (Todd, Waide, and Cornaby 1975). At Coweeta, NC, available nitrate and ammonium are rapidly immobilized by microflora or taken up by the root-mycorrhizal complex (Mitchell, Waide, and Todd 1975).

The functions of mycorrhizae to facilitate uptake of nutrients are described in Section 3.1.3. Mycorrhizal development is favored by low to moderate soil fertility (Pritchett 1979). In the eastern deciduous forest mycorrhizae are especially necessary for the growth of trees during drought or for species such as maples located on low-nutrient soils (Yahner 1995). Because high light intensity (or the resulting warm temperatures) favor mycorrhizae (Pritchett 1979), it would be expected that the organisms would not do well under the heavy canopies of old-growth deciduous forests in the spring or summer.

1.3.3.4 Vegetation

The eastern deciduous forest is defined by the dominance of angiosperm tree species that shed their leaves in fall. This forest type is species-rich, containing over 110 tree species (Yahner 1995). Some tree genera typical of eastern deciduous forests include oak (*Quercus spp.*), maple (*Acer spp.*), cherry (*Prunus spp.*), hickory (*Carya spp.*), and birch (*Betula spp.*). This forest ecosystem may be divided into four regions: northern hardwoods, central hardwoods, southern oak-pine, and bottomland hardwoods. Species typical of the four eastern deciduous forest regions are listed in Yahner (1995).

Eastern deciduous tree species may require soils of particular texture and nutrient content. For example, numerous trees such as maples, black cherry, and tulip (yellow) poplar require loamy soil (Yahner 1995).

The Hubbard Brook Experimental Forest (HBEF) in New Hampshire was chosen as a representative eastern deciduous forest location. This location is within the northern hardwoods region. Fourteen tree species, 11 shrubs, and 71 herb species are identified at HBEF. The dominant overstory species include sugar maple (*Acer saccharum* F.), American beech (*Fagus grandifolia* F.), and yellow birch (*Betula alleghaniensis* Marsh.) (Bormann and Likens 1979). The wide variety of understory species are described in Siccama et al. (1970).

1.3.3.5 Wildlife

Wildlife species diversity in eastern deciduous forests is high. These forests provide habitat for over 200 vertebrate species; approximately 17 amphibians, 23 reptiles, 154 birds, and 43 mammals (Yahner 1995). Diversity of reptile and amphibian species is greatest in the southern portions of this ecosystem, while that for birds and mammals is greatest in the northern portion (Yahner 1995). Representative wildlife species in the deciduous forest include white-tailed deer, red fox, wild turkey, box turtle, short-tailed shrew, downy woodpecker, and migratory warblers.

Diversity of wildlife within deciduous forests varies with stand age. Diversity is initially high in regenerating stands, declines in pole stands, then reaches a maximum in mature stands (DeGraaf et al. 1992). A detailed listing of species occurrences and habitat preferences for wildlife in northern hardwood forests are presented by DeGraaf et al. (1992).

1.3.3.6 Management practices

Eastern deciduous forests are managed for timber production, wildlife habitat, and recreation and aesthetics. They are not managed at the level of loblolly pine plantations and Douglas-fir forests. Timber management activities may include clearcutting, thinning, or setting prescribed fires. Unlike in the loblolly pine and Douglas-fir forests, fertilization is only used in experimental contexts.

Selective cutting permits forests to be managed for both timber and aesthetics (Yahner 1995). The shelterwood method of timber harvesting, a sequential cutting of mature trees over a 10-year period, is described in detail in Yahner (1995). This management practice is common throughout the eastern deciduous forest, especially as a method to regenerate oaks.

Many tree species in eastern deciduous forests (listed in Yahner 1995) are able to regenerate after clearcutting. The regeneration of particular species is dependent on whether seedlings have germinated and other seeds are in the soil. Herbaceous plants may be less resilient during clearcuts of old-growth forests (Duffy and Meier 1992). The cutting of single trees from a stand of intermediate-aged or mature trees benefits seedlings that prefer low light beneath the canopy, including sugar maple, American beech, and eastern hemlock. Cutting groups of trees is better for seedlings that grow well under only partial shade, including black cherry, tulip poplar, and white ash.

Other ecological effects resulting from thinning and clearcutting have been described. It has been suggested that the forest floor of northeastern hardwood forests decreases in depth for ten or twenty years after clear-cutting, even after the area is revegetated Bormann and Likens 1979). The low forest floor mass may continue to be observed for 60 to 80 years after cutting (Covington 1981). Nutrients are lost from forests, following cutting. The number of nitrifying bacteria has been shown to increase at Hubbard Brook and Coweeta, NC following the nitrogen losses associated with clearcutting (Waide et al. 1988). Whole-tree harvesting of trees that accumulate calcium, e. g., oak-hickory forests, may remove significant quantities of the element (Johnson and Henderson 1989). In Hubbard Brook, potassium is the element that exhibits the slowest recovery after clearcutting (Likens et al. 1994). With respect to wildlife, the removal of large oaks and hickories may deprive birds and mammals of mast (nuts and acorns). Similarly, numerous bird species rely on tree cavities for nesting locations (Yahner 1995).

Prescribed fires are used only occasionally in eastern deciduous forests either following timber removal or as a prerequisite to planting species that require bare soil (Lorimer 1990). The germination of aspen root suckers and oak acorns is aided by fire (Yahner 1995) and the thick bark of adult oaks make them fire-tolerant, though maple trees may exhibit high mortality. Habitat for ruffed grouse in the northern states or northern bobwhite in the southern states may also be managed through the use of fire (Yahner 1995). Fire volatilizes nitrogen in litter, though microbial activity may be increased in regions where substantial nitrogen has been immobilized in thick humic layers (Pritchett 1979).

1.3.4 Southwestern Semiarid Rangeland

1.3.4.1 Geography, climate, and soil classification

Semiarid rangeland (grassland and shrubland) is a dominant ecosystem type throughout much of the southwest, including portions of New Mexico, Colorado, and western Texas. The risk assessment for this region will focus on two sites in New Mexico, the Rio Puerco Resource Area and the Sevilleta National Wildlife Refuge, for which there are extensive environmental data, including experimental applications of municipal sewage sludge by the U. S. Forest Service.

The Rio Puerco Resource Area, located at the southeastern edge of the San Juan Basin, is a geological feature within the southeastern portion of the Colorado Plateau. The Nacimiento Mountains, with heights up to 9000 feet, bound the region on the east (Aguilar 1993). The soil classification for the sandy loam surface soil is found in Aguilar (1993). The soil is deep and well-drained, with ground water depths estimated at 200 to 300 feet for higher elevations and somewhat less for lower ones. References for the permeability rate, infiltration rate, and leaching potential are available in Aguilar (1993). Information about wind direction and frequency of calm can also be found in Aguilar (1993). The area is administered by the US Department of the Interior Bureau of Land Management. The sludge application demonstration site is used mainly for livestock grazing and wildlife. A comprehensive description of water and land use in the Rio Grande Valley is found in Ellis et al. (1993). At Cuba, NM, the nearest NOAA station to the Rio Puerco Resource Area, the average temperature in January is -3.5°C and in July is 20.4 °C.

The Sevilleta National Wildlife Refuge, located south of Albuquerque, NM, is administered by the United States Department of Interior. This Long-Term Ecological Research Site (LTER) receives funding from the National Science Foundation (Aguilar et al. 1994). Additional data will be obtained from 1) the Sierra Blanca Ranch in the Chihuahuan Desert, operated by MERCO Joint Venture using New York City municipal sludge, and used for research purposes by Texas Tech University (Sosebee 1994) and 2) a semiarid sagebrush ecosystem in Wolcott, CO, where Fort Collins municipal sewage sludge is applied experimentally by researchers at Colorado State University (Harris-Pierce 1994, Pierce 1994).

Precipitation is the most important determinant of plant communities and productivity for semi-arid regions. Where precipitation is below 500 mm per year, forage production increases substantially with precipitation (Holechek et al. 1989); where rainfall is above 500 mm per year, soil characteristics are more important for production. The region receiving these limiting quantities of precipitation include 80% of the area in the 11 westernmost states of the U. S (Holechek et al. 1989). Annual precipitation at Rio Puerco and Sevilleta sites is approximately

250 mm (Aguilar 1993, Aguilar et al. 1994), with a large percentage occurring in high intensity, convectional thunderstorms that are not ideal for sustained forage growth. Precipitation in the throughout areas of the southwestern United States (e.g., Arizona) is bimodal, with a general pattern of winter precipitation, spring drought, summer precipitation, and fall drought. Not only the amount of precipitation, but the distribution, the relative humidity, the form of precipitation and annual variability (e.g., droughts) influence vegetation growth (Holechek et al. 1989). For example, range forage reproduction is tied to summer precipitation. Wind can increase moisture losses to evaporation and increase plant transpiration. Holechek et al. (1989) provide a table of average annual precipitation and forage production for 23 range locations, including South-central New Mexico (desert grassland), Northeastern Colorado (shortgrass prairie), and Northwest Texas (southern mixed prairie).

The most important property of soils in any semiarid region is low organic matter (Loftin et al. 1995a). Inputs to the low organic matter fraction of soil (<1% to 3%) include surface litter and net root production. Aguilar and Heil (1988) have found that organic matter in the Northern Great Plains rangeland, including carbon, nitrogen, and phosphorus, is associated with microclimate, landscape position, and soil parent material. The dependence probably applies to southwestern and western rangeland as well. Loftin et al. (1995a) provide references for the soil aggregate stability and resistance to erosion provided by increased organic matter content of soil. Soils at the Sevilleta National Wildlife Refuge are mapped as the Harvey-Dean Association: deep and well-drained Calciorthid soils derived primarily from sandstone and limestone (Aguilar and Loftin 1994).

Soil erosion has been judged to present the greatest risk to surface water quality in the Rio Grande Basin (Ellis et al. 1993). Water-induced soil erosion is more important in semiarid areas, and wind-induced erosion is more important in arid areas (Fletcher et al. 1978). The process is closely connected to individual thunderstorm or snowmelt events (Fletcher et al. 1978). Widespread storms can cut arroyos and lower water tables, leading to desertification of heavily grazed grasslands (Fletcher et al. 1978). In the Middle Rio Grande Basin, quantities of runoff and sediment may be controlled by vegetation, ground cover, infiltration rates, and other soil properties (Loftin et al. 1995a). For example, two significantly different plant communities have been found a few hundred meters apart in the Rio Puerco Watershed Resource Area, with runoff rates that differ by a factor of 6 to 8 and sediment yields that differ by a factor of 10 to 15.

1.3.4.2 Nutrients

In rangeland soils, nitrogen is usually the most limiting nutrient, followed by phosphorus and perhaps potassium and sulfur. Grassland soils generally have C:N ratios between 14 and 20 (Fletcher et al. 1978). The mechanisms of transfer of nitrogen in and out of rangeland ecosystems include: deposition of particulates, aerosols, and NH_3; symbiotic and nonsymbiotic nitrogen fixation, removal of nitrogen by grazing animals, loss of NH_3 to the atmosphere, and the transport of nitrogen material by wind (Woodmansee 1978).

Moisture-dependent nitrogen fixation at the surface of arid and semiarid soils and litter and microbial accumulation of nutrients can result in the accumulation of nitrogen at the soil surface, with subsurface horizons remaining depleted of the nutrient. However, nodulation of legumes is not a common mechanism for the process. Farnsworth et al. (1978) have suggested that free-living microorganisms such as *Azotobacter* sp. or *Clostridium* sp. or mycorrhizae may be responsible for nitrogen fixation in deserts. In contrast, Woodmansee (1978) cites several

studies that show little or no nitrogen fixation under typical grassland conditions. Under these conditions, the top few cm of soil may have lower nitrogen content and fewer roots than deeper surface layers.

The translocation of nitrogen by erosion is significant (Fletcher et al. 1978). Woodmansee (1978) suggests that large, domestic ungulates are important vectors of nitrogen loss, with 50% of the excreted nitrogen lost as NH_3. The net loss depends on the grazing intensity.

Productivity in rangeland ecosystems is generally measured as the annual production of herbage and browse per acre. The average annual desert grassland production is 307 pounds per acre (air-dried; USFS 1981). As discussed above, soil water content is generally more limiting than nutrients, though when moisture is adequate, grass growth responds more to nitrogen than to added water.

1.3.4.3 Soil biota and processes

Termites and ants are common in desert soils, but their role in soil development is largely unknown (Fuller 1974). Termites move soil into mounds and create channels in soil. Their presence necessitates the presence of cellulose-hydrolyzing protozoa.

1.3.4.4 Vegetation

The arid and semiarid rangelands of the Western United States are dominated by a complex of bunchgrasses, annual grasses, cacti, salt-tolerant shrubs, sagebrush, pinyon-juniper, and chaparral (USFS 1981). Multiple vegetation assemblages that include scrub and shrub lands and desert and plains grasslands have been identified within this ecosystem (Loftin et al. 1995a).

Vegetation within the Rio Puerco watershed, a representative semiarid rangeland location in New Mexico to which sludge has been applied, consists of semiarid grasses and shrubs (Aguilar 1993). Dominant grass species include blue grama, ring muhly, galleta, sand dropseed, bottlebrush squirreltail, needlegrass, and western wheatgrass. Common shrubs include fourwing saltbush, winterfat, and broom snakeweed. The Sevilleta National Wildlife Refuge, located in the Rio Grande Valley of New Mexico, is dominated by semiarid shortgrass prairie and great basin shrubland at low elevations and pinyon juniper woodland at higher elevations (Loftin and Aguilar 1994). One experimental community included blue grama, hairy grama, and black grama, ring muhly three awn, the forb yellow spiny-aster, and the shrubs narrow-leaf yucca, winterfat, four-wing saltbush, and groundsel (Loftin and Aguilar 1994). Plant communities have shifted in the last century because of a combination of past grazing practices, changing fire frequency, and climatic change (Bahre 1993). For example, decreases in fire frequency due to overgrazing have caused increases in fire intolerant species in the plant communities (Loftin et al. 1995a).

Plant growth and production are generally determined by soil water content (Loftin et al. 1995a). Other determinants of the species and quantity of vegetation in a semiarid rangeland area are measures of climate and topography, as well as soil characteristics, including color, texture, structure, depth, pH, and organic matter and mineral content (Holechek et al. 1989). Although clay soils retain nutrients better than sandy soils, sandy soils in the arid west are generally more productive than clay soils because the former facilitate infiltration rather than

runoff of precipitation (Holechek et al. 1989). Additionally, in coarser-textured soil, water is generally more available than in finer-textured soils.

1.3.4.5 Wildlife

A wide range of wildlife species use arid and semiarid rangeland. Large mammals may include mule deer, pronghorn, skunks, badger, raccoon, fox, and coyote (Aguilar 1993). Rodents, such as kangaroo rats, ground squirrels, pocket and harvest mice, prairie dogs, and pocket gophers are common. Other small mammals that may be present include the desert shrew. Several species of lizards and snakes (rattlesnakes in particular) may be common and abundant. Many birds may use this habitat, but because they are highly mobile and commonly migratory, few are permanent residents (Aguilar 1993).

1.3.4.6 Management practices

Most of the rangelands in the U. S. have been grazed for at least 100 years, and perhaps as long as 450 years (USFS 1981). An unmanaged system exists only in a historical context. The term range condition is used to describe "the degree to which the present vegetation and ground cover depart from that which is presumed to be the natural potential (or climax) for the site" (USFS 1981). The range condition is a comparison of the estimated kinds and quantity of vegetation that would exist in the absence of livestock, given the existing climate, physiography, and soils, to current parameters (USFS 1981). The condition of rangeland in the southernmost States is poor, with less than 40 percent of the rangelands in California, Arizona, New Mexico, and Texas in good or fair condition (USFS 1981).

Numerous factors affect the management goals and practices of rangeland. These include: climate, weather, topography, primary production, markets for products, and goals and finances of the owner (USFS 1981). Management levels (e.g., no livestock, intensive management) vary, depending on the number of livestock and whether the only management goal is to maximize production. According to the U.S. Forest Service (1981), management tools may include: "1) grazing management including kinds and classes of livestock, stocking rates, grazing seasons and improved systems of grazing and 2) range improvements including water development, fencing, seeding, and undesirable plant control and pest management and control using mechanical, fire, chemical, and biological methods." The use of fertilizers on rangeland has been considered but rarely used in the past (USFS 1981). Among rangeland sites, the Sevilleta National Wildlife Refuge is less managed than others.

Range management practices that are intended to improve livestock grazing may also improve habitat for wildlife. Fences provide perches for birds and small mammals and are also used to control hunting (USFS 1981). Populations of upland game birds may increase near water developments, range seedings, and prescribed range burns (USFS 1981).

1.4 ASSESSMENT AND MEASUREMENT ENDPOINTS

Two categories of ecological endpoints, assessment and measurement endpoints, are identified in preparation for the risk assessment. An assessment endpoint is completely defined if it includes an entity (e.g., individual, population, community, ecosystem); a property of that entity to be protected (e.g., growth, reproduction, survival); a level of effect (relative to reference locations) (see Sect. 1.6.1); and a desired statistical confidence level (Suter et al. 1995). The

statistical confidence level is not defined in this assessment because of the wide range of studies and numerous extrapolations among organisms and ecosystems. Measurement endpoints are quantitative or qualitative summaries of a measurement or series of measurements that are related to effects on an assessment endpoint (Suter 1989, EPA 1992b).

Assessment endpoints are chosen on the basis of ecological importance, policy or societal significance, susceptibility to toxic effects, and appropriateness of the level of organization to the spatial scale. The level of organization of an endpoint is appropriate to the spatial scale of a site if toxic effects on the site could be significant for the endpoint . In general, the spatial boundaries of a contaminated site are defined prior to the initiation of a risk assessment, usually based on institutional rather than habitat considerations. In the current assessment, flexibility is needed to accommodate the range of scales associated with non-agricultural lands on which sludge may be applied. The size of sites treated with sewage sludge may range from a single ha to many hundreds of ha. Thus, in the context of this assessment, spatial scale is not precisely defined. Spatial scale is particularly critical to selecting endpoints and arriving at conclusions in a wildlife risk assessment. If risks to individual animals that use sludge-treated sites are expected, the size of a treatment plot determines whether such impacts are significant at the population level of organization.

Assessment endpoints for the soil, plant, and terrestrial animal subsystems are presented in Tables 1.1, 1.2, and 1.3, respectively. The distinction between the subsystems is discussed below in Sect. 1.5, Conceptual Models. Some assessment endpoints represent entities that are clearly valued, such as populations of shrews. In contrast, other assessment endpoints represent entities that are important to the function of an ecosystem, but there is no agreement about which (if any) directional change constitutes a risk. The lack of agreement may arise from 1) differing management goals in all of the ecosystems for which the endpoint is relevant, 2) stakeholders with various interests in a particular ecosystem, or 3) the lack of knowledge of long-term impacts of a change in an ecosystem function, such as decomposition rate. These endpoints are marked in the tables. Guidance from EPA regarding the selection of assessment endpoints may be forthcoming in the agency's *Guidelines for Ecological Risk Assessment*. Proposed guidelines were published in September 1996 (EPA 1996).

Table 1.1. Assessment and Measurement Endpoints for the Soil Subsystem

Assessment Endpoint	Measurement Endpoint
Change in relative abundance of taxonomic groups of soil invertebrates (E)	1) Field surveys of soil invertebrate communities
Reduction in abundance or growth of soil invertebrates (R)	1) Single-chemical toxicity 2) Toxicity tests with sludge-treated soil 3) Field surveys of soil invertebrates
Change in microbial biomass (E)	1) Field measurements of microbial biomass
Change in microbial diversity (E)	1) Field surveys of microorganisms
Change in decomposition (rate or equilibrium extent) of native soil organic matter and plant litter (E)	1) Single-chemical toxicity 2) Tests with sludge-treated soil
Change in balance of processes in nitrogen cycle (E)	1) Single-chemical toxicity 2) Toxicity tests with sludge-treated soil
Change in extent of colonization of trees or other plants by vesicular-arbuscular mycorrhizal fungi (E)	1) Field surveys of VAM fungi
Increase in erosion (R)	1) Net drainage determined in water budget 2) Field observations
Leaching of nutrients or toxicants (R)	1) Net drainage determined in water budget 2) Field observations
Surface runoff of nutrients or toxicants (R)	1) Net drainage determined in water budget 2) Field observations

R: A positive determination means that risk to the endpoint is likely.
E: A positive determination means that an effect is likely, but whether the effect constitutes a risk depends on regulatory values, management goals, and other site-specific characteristics.

Table 1.2. Assessment and Measurement Endpoints for the Plant Subsystem

Assessment Endpoints	Measurement Endpoints
Reduction in growth, yield, or survival of plant populations (R)	1) Single-chemical toxicity 2) Toxicity tests with sludge-treated soil
Increase in production of dominant forest trees (E)	1) Modeled estimate of production using nitrogen in sludge-treated soil 2) Vegetation surveys
Change in relative dominance of forest trees (E)	1) Modeled estimates of species production using nitrogen in sludge-treated soil
Change in herbaceous biomass (E)	1) Vegetation survey
Change in forest understory composition (E)	1) Vegetation survey
Change in species richness and diversity (E)	1) Vegetation survey
Increase in nutrient content of plants (E)	1) Vegetation survey

R: A positive determination means that risk to the endpoint is likely.
E: A positive determination means that an effect is likely, but that whether the effect constitutes a risk depends on regulatory values, management goals, and other site-specific characteristics.

Table 1.3. Assessment and Measurement Endpoints for the Terrestrial Animal Subsystem

Assessment Endpoints	Measurement Endpoints
Reduction in abundance or reproduction of populations of mammalian ground invertebrate feeders (shrews) (R)	1) Single-chemical toxicity 2) Biological surveys, including organ concentrations of toxicants 3) Toxicity tests with sludge-treated soil
Reduction in abundance or reproduction of individual avian invertebrate feeders (robins or meadowlarks) (R)	1) Single-chemical toxicity 2) Biological surveys, including organ concentrations of toxicants
Reduction in abundance or reproduction of individual large herbivores (deer) (R)	1) Single-chemical toxicity 2) Biological surveys, including organ concentrations of toxicants
Reduction in abundance or reproduction of individual predators (fox) (R)	1) Single-chemical toxicity 2) Biological surveys

R: A positive determination means that risk to the endpoint is likely.
E: A positive determination means that an effect is likely, but that whether the effect constitutes a risk depends on regulatory values, management goals, and other site-specific characteristics.

1.5 CONCEPTUAL MODELS

Conceptual models graphically represent working hypotheses of the linkages between a stressor and assessment endpoints (EPA 1992b). The processes in such models include the transport of contaminants or other stressors within the site, the movement of contaminants or other stressors away from the site, the uptake by biota (either directly or through food webs), and propagation of secondary effects through ecological interactions. The conceptual model may either: 1) provide a rather exhaustive picture of transfers of stressors in an ecosystem or 2) focus the risk assessment on the most important endpoint-stressor relationships, for which data are available. The conceptual models for this assessment are in the former category, since they were developed by with the peer review team for the project before final assessment endpoints were selected and before data were collected.

Preliminary conceptual models and assessment endpoints for evaluating the ecological effects of the land application of sludge were developed prior to the workshop held in Oak Ridge, TN in April 1995. Models described below represent the consensus of the workshop participants and include effects from both contaminants and nutrients in sludge. Assessment endpoints were chosen from the domains in the conceptual models. The focus of this assessment is on terrestrial impacts. Consequently, aquatic pathways and endpoints were not considered. Information on erosion, runoff, and leaching necessary to begin an aquatic assessment may be found in Sect. 3.

The generic model of a terrestrial ecosystem to which sludge is applied was broken into three subsystems: soil, plant, and terrestrial animal subsystems (Fig. 1.1). These subsystems are linked through the transfer of contaminants and nutrients and by modification of habitat. Detailed descriptions of the components within each subsystem, the interactions among subsystems, and assessment and measurement endpoints for each subsystem are outlined below.

It was assumed that, in general, the types of ecosystem processes (if not the relative importance) are comparable among the four ecosystems for which impacts of sludge application are being assessed. Therefore, ecosystem-specific conceptual models were not developed. To evaluate risks within each ecosystem, parameters and data specific for that ecosystem were used. In the absence of suitable ecosystem-specific data, substitutes from similar ecosystems were used.

1.5.1 Soil Subsystem

Components and attributes of the soil subsystem and linkages to other subsystems are graphically displayed in Fig. 1.2. The addition of municipal sewage sludge may affect both biotic and abiotic soil processes, resulting in altered bioavailability, transfer, and flux of nutrients and contaminants. The arrows in the figure represent either the direct transfer of contaminants and nutrients or other interactions among components within the soil subsystem.

Sludge application may alter the physical and chemical characteristics of soil (e.g., structure, water retention, bulk density, organic carbon, cation exchange capacity) and may change the quantity and quality of inorganic nutrients, carbon, and contaminants that are available for uptake by soil organisms. These changes are likely to have impacts on the microbial, invertebrate, and plant communities.

Contaminants and nutrients added through sludge may exit the soil subsystem through the air or water or through biomass, such as mobile wildlife or harvested plants. Metals and nitrogen (as nitrate) may be transported to surface water or leach to groundwater. Ammonia from sludge may be released to the atmosphere. While sludge application may increase the amount of carbon dioxide (the primary product of decomposition) released to the atmosphere, the amendment may reduce net CO_2 emissions by promoting increased production of plant biomass and associated CO_2 fixation. The erosion of soil may be reduced by the addition of sewage sludge, though the construction of access roads in forests could increase erosion.

The microbial community is responsible for numerous processes, including decomposition, nitrogen fixation, nitrification, biodegradation of contaminants, and the facilitation of nutrient uptake by plants. The addition of sludge may affect each of these functions. The application of sludge may also introduce microorganisms to the soil (e.g., human pathogens or nitrifying bacteria). These organism introductions are not addressed in this report. Soil invertebrates may be affected by sludge application through increased resources, contaminant exposure and changes in soil structure. Soil invertebrates may also serve as a contaminant transfer pathway to the terrestrial animal subsystem. Assessment and measurement endpoints for soil subsystem are presented in Table 1.1.

1.5.2 Plant Subsystem.

Components and attributes of the plant subsystem and linkages to other subsystems are graphically displayed in Fig. 1.3. Sludge amendments may result in alterations to the total plant biomass and possibly the species composition of treated areas. As a result, nutritional quality of the plants for herbivorous animals may be altered. Nutrients and contaminants in the soil may also be taken up by plants. These contaminants may be directly toxic to the plants or they may be transferred to insects or vertebrate herbivores that feed on the plants. Due to differential plant growth and reproduction responses and toxicity, the species composition and distribution within the plant community may be altered. Sludge-mediated changes in the presence or abundance of soil-associated plant pathogens and pests (e.g., nematodes) or soil structure and function (e.g., water balance) may affect plant growth or survivorship. Habitat properties, such as percent cover and height of plants may be affected. Lastly, changes in growth and nutritional composition of plants may alter the production and quality of litter that is returned to the soil subsystem. Assessment and measurement endpoints for the plant subsystem are presented in Table 1.2. This assessment will address the importance and likelihood of occurrence of the pathways in the conceptual model.

1.5.3 Terrestrial Animal Subsystem.

Components and attributes of the terrestrial animal subsystem and linkages to other subsystems are graphically displayed in Fig. 1.4. These represent theoretical pathways whereby contaminants or other stressors may be transferred from sludge to soil to terrestrial biota. Terrestrial vertebrates may be exposed to and affected by contaminants in sludge. Exposure may be through the food web (e.g., soil-plant-herbivore-predator; soil-plant-insect-insectivore-predator; soil-earthworm-vermivore-predator (vermivores are organisms that feed on worms)) or through incidental ingestion of soil while foraging or grooming.

Nutrient-mediated changes in the plant community may alter habitat structure at sludge-treated sites, making the site more or less suitable for various animal species. Increased quantity

or quality of plant biomass may increase the abundance of herbivores, such as insects. Increased abundance of some insect species may adversely affect the growth or production of their plant hosts. In addition, due to the close association of plants and insects, changes in the species composition of the plant community will result in changes in the species composition of the insect community. Contaminants in plants may also be toxic to insects or may be accumulated and transferred to vertebrate insectivores.

Assessment and measurement endpoints for terrestrial animal subsystem are presented in Table 1.3. Trophic groups for which risks will be assessed include herbivores, vermivores, insectivores, and carnivores. Because the species composition of the terrestrial animal community differs among ecosystems, representative taxa for each trophic group were selected for each ecosystem (Table 1.4).

Table 1.4. Representative, ecosystem-specific wildlife endpoints

Trophic Group	Ecosystem			
	Northwestern Douglas-fir Forest	Semi-arid Rangeland	Southern Pine Plantation	Eastern Deciduous Forest
Herbivore	Black-tailed Deer	Mule Deer	White-tailed Deer	White-tailed Deer
Mammalian ground invertebrate feeder	Trowbridge Shrew	Desert Shrew	Short-tailed Shrew	Short-tailed Shrew
Avian invertebrate feeder	American robin	Western meadowlark	American robin	American robin
Predator	Red Fox	Kit Fox	Red Fox	Red Fox

1.6 ASSESSMENT APPROACH

1.6.1 Contaminants

A primary concern related to the land application of sewage sludge is the impact of organic and inorganic contaminants that may be present at the completion of the treatment process. If concentrations are sufficiently high, these contaminants in sludge-treated soil may present a hazard to terrestrial plants, wildlife, soil invertebrates, microorganisms, or ecosystem-level processes controlled by these organisms.

A weight-of-evidence approach (EPA 1992b) is employed to assess risks from contaminant exposure. This approach uses multiple lines of evidence to estimate the likelihood that significant effects are occurring or will occur and to describe the nature, magnitude, and extent of effects on the designated assessment endpoints. The lines of evidence may include results from chemical analyses, toxicity tests, biological field surveys, and bioindicators, if available.

Where possible, a 20% level of effects on assessment endpoint properties (e.g., 20% difference in germination or growth compared to control plants) has been used. This effects level has been chosen for use in ecological risk assessments at the Oak Ridge Reservation, TN, and has various precedents in regulatory criteria, detection limits of field measurement techniques, and lowest-observed effects concentrations (e.g., for dietary tests of reproduction in birds) (Suter et al. 1995).

Chemical concentration data from the 1988 NSSS have been used to model contaminant exposure and to estimate risk.

1.6.1.1 Plants, soil invertebrates, and soil microbial processes

Lines of evidence (measurement endpoints) available to assess contaminant risks to assessment endpoints described in Table 1.3 include: chemical analyses (single-chemical toxicity), ambient media toxicity tests (using sewage sludge-treated soil), and biological field surveys.

The first line of evidence consists of comparisons of concentrations of chemicals shown to cause toxic effects on soil-associated biota (ecotoxicological benchmarks derived at Oak Ridge National Laboratory) to concentrations of contaminants in sludge-amended soil. Exposure levels were derived from the contaminant concentrations reported in the NSSS (Sect. 4.1). To be conservative, the 95% UCL around the mean for the lognormal distribution of each contaminant concentration in the NSSS was used. Concentrations of contaminants in sludge-amended soil were compared to toxicological benchmarks to estimate the presence, nature, and magnitude of risks to soil-associated biota. The assumptions regarding mixing of sludge and soil are presented in Sect. 4.1.1. Concentrations of contaminants in soil were compared to ecotoxicological benchmarks for plants (Efroymson et al. 1997a), soil invertebrates, and soil microbial processes (Efroymson et al. 1997b). The output of this initial conservative screening process for each ecosystem was a list of Contaminants of Potential Ecological Concern (COPECs).

Sludge in forest and rangeland ecosystems is not rapidly incorporated into soil, as it is with tilling and disking practices in agriculture; sludge is simply deposited on the soil surface (Sect. 1.2). Thus, a series of assumptions was made, including the asumption that municipal sewage sludge is distributed homogeneously through the top 15 cm of soil, an assumption that was made in the Technical Support document for 503. These assumptions are detailed in Section 4.1. Although the bioavailability of contaminants in sludge may decrease as contaminants become bound to organic matter (Chaney and Ryan 1994), contaminant-specific and soil-specific binding factors for COPECs in sludge are not available. After this report is completed, adjustments may be made using such factors as they are developed by David Brown at USEPA Environmental Research Laboratory in Athens, GA.

Ambient media toxicity tests represent a second line of evidence for the assessment of risks to plants and soil biota. These are laboratory tests evaluating growth, survival, or reproduction of soil-associated biota maintained in sludge-amended soil. Where data were available, these tests provided estimates of toxicity that incorporated combined toxic effects of all chemicals, realistic chemical speciation, and realistic bioavailability in applied municipal sewage sludge-treated soil. Rarely were these tests available for the particular ecosystems of this assessment

The last line of evidence that may contribute to an evaluation of risks to soil biota consists of field surveys. These are observations of the health, abundance, and diversity of soil-associated biota at sludge-amended sites. Differences between parameters measured at sludge-amended and non-amended sites may indicate effects from sludge application. It should be emphasized that observed effects can represent the interaction between nutrients and contaminants in sludge.

The weight of evidence takes into account the fact that contaminant concentrations in sludge-treated soil in toxicity tests and field surveys often differ from those using sludge containing the 95% UCL around the mean NSSS concentrations and that application rates are usually different from the rate assumed in this assessment (Sect. 4.1).

1.6.1.2 Wildlife

Lines of evidence available to assess contaminant risks to wildlife include: chemical analyses (single-chemical toxicity), biological field surveys, and toxicity tests with sludge-treated soil (available for shrews only). Each line of evidence was used to evaluate risks to the four wildlife trophic groups: herbivores, mammalian ground invertebrate feeders, avian invertebrate feeders, and predators (Table 1.3).

The first line of evidence, analyses of COPECs in sludge, was used to estimate contaminant exposure experienced by representative wildlife species in sludge-treated areas. Exposure was estimated using the generalized exposure model outlined in Sample and Suter (1994):

$$E_j = \sum_{i=1}^{m} \frac{IR_i \times C_{ij}}{BW}$$

where:

E_j	=	total estimated exposure to contaminant (j) (mg/kg/d)
m	=	total number of ingested media or potential routes of oral exposure (e.g., food, soil, etc.)
IR_i	=	ingestion rate for medium (i) (kg/d or L/d)
C_{ij}	=	concentration contaminant (j) in medium (i) (mg/kg or mg/L)
BW	=	body weight of endpoint species (kg)

Total estimated exposure included exposure through food and incidental ingestion of soil (with concentrations in sludge substituted for soil). Concentrations of COPECs in food items were estimated using contaminant-specific uptake factors from soil or sludge-treated soil to plants, earthworms, or wildlife. The derivation of the uptake factors is discussed in Sect.4.2.1.2. Exposure estimates generated from this model were compared to toxicological benchmarks for wildlife developed according to the methods outlined in Sample et al. (1996b).

Field surveys represent the second line of evidence for the assessment of risks to wildlife. As described above, these consist of observations of the health, abundance, and diversity of wildlife at sludge-amended sites. Differences between sludge-amended and non-amended sites may indicate effects of sludge application. Again, it should be noted that

observed effects cannot be attributed solely to contaminants and may represent the interaction between the nutrients and contaminants in sludge or effects of the nutrients alone.

Some studies of field applications of sludge provide data on biomarkers. These contribute to the second line of evidence. Biomarkers are biochemical or physiological changes which indicate that an organism has received an internal dose of a chemical. Examples include contaminant body burdens, histopathological observations, and measures of detoxification enzyme levels that may be induced under various conditions of stress. These data were used to the extent that they were available to better describe the nature and extent of risks.

The third line of evidence, available only for shrews, is laboratory toxicity tests of field-collected media . To evaluate uptake and transfer of heavy metals at sludge-treated sites to secondary consumers, shrews were fed diets containing earthworms collected from sludge-treated and untreated sites.

The weight of evidence considered the fact that contaminant concentrations in single-chemical toxicity tests, field surveys, and biomarkers differed from those in the NSSS. Treatment studies with application rates similar to the rate assumed in this assessment were given high weight in the analysis.

1.6.2 Nutrients

The nutrient components of sludge (primarily nitrogen, but perhaps phosphorus or potassium) may impact the growth and production of plants, forest community structure, habitat quality for wildlife, forage quality for wildlife, decomposition and nitrogen cycling, and leaching of nutrients such as nitrate. These assessment endpoints and the linkages among them are described in the conceptual models (Sect. 1.4). The approach to evaluating impacts of nutrients was both qualitative and quantitative, and is described below. This risk assessment assumed that municipal sewage sludge was the only fertilizer applied; thus the baseline condition was no fertilization.

1.6.2.1 Water budgets

To adequately evaluate the status of leaching, erosion, tree production and other assessment endpoints, water budgets for each ecosystem were defined. Runoff and drainage in Douglas-fir forests, loblolly pine plantations, eastern deciduous forests were estimated from precipitation, evapotranspiration, and soil water storage. Monthly average precipitation values were gathered for sites in the three forest ecosystems, and potential evapotranspiration was estimated from a climate atlas of the United States. Monthly drainage obtained from water budget analyses was used to estimate leaching potential. Precipitation data was obtained from individual sludge application sites or from nearby stations recorded in the U. S. Historical Climatology Network (NOAA). Soil water storage data was obtained from the State Soil Geographic (STATSGO) database, developed by the U. S. Natural Resources Conservation Service. Mean nitrogen (Kjeldahl, nitrite and nitrate) concentrations in sewage sludge were calculated from the NSSS. Predictions of leaching were compared to observed leaching at sludge study sites in the ecosystems.

1.6.2.2 Erosion

The potential for increased plant biomass following sludge application may decrease erosion in ecosystems with relatively low plant biomass, such as the semiarid rangeland. This issue was examined using data from sludge application research sites in New Mexico and Sierra Blanca Ranch in the Chihuahuan Desert, Texas.

Because of the high standing biomass, large changes in erosion patterns following sludge application were not expected in forested ecosystems, except possibly during early regrowth after clear-cutting. Thus, we did not examine erosion as an assessment endpoint for established Douglas-fir forest, eastern deciduous forest, or the southeastern pine plantations. The largest erosion risk associated with application of sludge to forest land is probably from vehicle access roads constructed for sludge application and is dependent on road construction, vehicle tires, and the induction of convergent flow. Erosion increases as a power function of water flow rate.

1.6.2.3 Plant productivity and community structure

Nutrient constituents and increased soil water storage associated with sewage sludge can have significant impacts on plant growth and productivity, the composition of the forest floor, and nutrient transformations. A model called LINKAGES has been developed to examine the relationships between nitrogen cycling and long-term forest stand dynamics, limited by climate and soil water status (Pastor and Post 1985). LINKAGES can predict: soil nitrogen and water availability with vegetation response, carbon and nitrogen stored in the forest floor, soil respiration, and carbon to nitrogen ratios in soil (Pastor and Post 1985). The range of possible outputs includes:

1) number of stems
2) aboveground biomass
3) leaf litter
4) total litter
5) total net aboveground production
6) available nitrogen
7) humus C:N ratio
8) soil organic matter
9) evapotranspiration
10) individual species biomass
11) number of dead stems
12) basal area of dead trees

The model simulates the birth, growth, and death of individual trees greater than 1.43 cm diameter at breast height on a circular 1/12 ha plot. Species reproduction and tree growth are constrained by degree days, light and water availability, and tree growth is also constrained by nitrogen availability. Thus, competition among species is determined by the variable responses of the species. Incoming sunlight is the variable which drives the model. The structure of the model, including site conditions, predicted tree growth, and population dynamics are described in Pastor and Post (1985).

The following site data are inputs to the model, taken directly from Pastor and Post (1985):

1. latitude for making sun-angle corrections

2. days of the year the growing season begins and ends (last and first killing frost)
3. monthly mean temperatures (¡C) and precipitation (cm) and their standard deviations
4. field capacity and wilting point of the soil profile with roots in cm [a figure with these quantities relative to soil texture appears in Pastor and Post (1985)]
5. initial soil organic matter and nitrogen contents (Mg ha^{-1})

Organic and nitrogen components of sewage sludge may be treated as initial soil concentrations or as subsequent additions to the site, as the model treats nitrogen in acid precipitation.

LINKAGES was adapted to the eastern deciduous forest at Hubbard Brook and to Douglas-fir and loblolly pine plantations. Nutrient cycling and plant community endpoints for the semiarid rangeland were determined based on existing study site literature. Grassland ecosystem models such as the CENTURY model (Parton et al. 1992) may be used for modeling of sludge additions to rangeland in the future.

LINKAGES was used to examine ecosystem -level (production) effects of sludge application in the loblolly pine plantations, eastern deciduous forest, and Douglas-fir forest. Community-level changes were modeled in the eastern deciduous forest.

The output from LINKAGES was very different from the usual output of an ecological risk assessment, such as that described above for metal and organic contaminants. In most risk assessments contaminant exposure levels in wildlife, plants, or other assessment endpoints are compared to known benchmarks of ecotoxicological significance. Significant risks from the contaminant are predicted if its concentration exceeds these values. Because the nutrient-generated types of changes in an ecosystem that actually constitute "risks" are not agreed upon, we could only perform a statistical comparison of alternative scenarios using uncertainty analyses (e.g., dominant trees prior to and after the application of sewage sludge) and look for thresholds for productivity and/or species losses. PRISM is an uncertainty analysis and sensitivity analysis tool (Sect. 2.5) that can be used, for example, to rank parameters affecting productivity.

Where possible, empirical data on changes in plant community structure and productivity were obtained from municipal sewage sludge application study sites. These were included in Sect. 4.4, the contaminant Section rather than in Sect 2, the nutrient modeling section.

2. IMPACTS OF NITROGEN

2.1 SIMULATION METHODS

The LINKAGES model (Pastor and Post 1985) was selected for evaluation of biosolids effects on forest productivity and community structure. LINKAGES is a general model of carbon and nitrogen fluxes of forest ecosystems for exploration of relationships between stand dynamics and nutrient cycling over time scales up to several centuries. The model simulates the annual establishment, growth, and mortality of all trees with a diameter at breast height greater than 1.43 cm located within a $^1/_{12}$ ha plot. Under favorable conditions, the tree diameter is incremented annually according to a function that produces $^2/_3$ of the diameter growth at $^1/_2$ of the maximum tree age. Changes in litter carbon and nitrogen contents are simulated with algorithms for decomposition of litter (leaf, wood) cohorts of differing ages. Decomposition is influenced by litter quality (C/N ratio), temperature, and water availability (Fig. 2.1). Nitrogen released in decomposition influences tree growth if soil water availability or light are not more limiting growth factors.

Each biosolids application was added to the dry matter and nitrogen components of the humus pool in four sequential annual increments of 30, 6, 3, and 2% of the application treatment. These increments were based on observations of changing biosolids decomposition rates with time (Keeney et al. 1975, Lindemann and Cardenas 1984, Pratt et al. 1973, Sims and Boswell 1980). The mean Kjeldahl nitrogen concentration from the National Sewage Sludge Survey of 45.9 g/kg was used and the carbon content of sludge was assumed to be 50% of dry weight to give a mean C/N ratio of 10.9. The nitrite and nitrate content of a biosolids treatment was added directly to the available N pool during the year of application only (Fig. 2.1). The remainder of the carbon and nitrogen in the biosolids (59%) was considered to be lost by volatilization or released as carbon dioxide and nitrogen gases (Varanka et al. 1976, Terry et al. 1979). Decomposition of humus proceeds at an annual rate of 50 mg/g (5%) in LINKAGES providing increments of mineral nitrogen into the available N pool which affects tree growth and reduces soil organic matter. Annual additions of N and organic matter to the humus pool are derived from the aging litter cohorts. A description of algorithms for litter decay and the effect of available soil N on tree growth is given by Pastor and Post (1985).

A sensitivity analysis of the input parameters of LINKAGES has been conducted and this produced a ranking of the parameters having significant influence on tree growth. An important soil property is the available water content, defined as the difference between field capacity and wilting point for the root zone depth. Higher available soil water favors tree growth. Soil organic matter content is also a sensitive parameter which has its effect on growth through effects on nutrient cycling. Input values for the carbon/nitrogen ratios of litter and soil organic matter also influence decomposition and nutrient cycling and are sensitive parameters.

2.2 MODEL INPUTS

The site-specific input parameters required for simulation (Table 2.1) show a range of latitudes, longitudes, growing seasons, and soil water storages (determined by the difference in field capacity and wilting point). Stochastic climate conditions used in LINKAGES simulations were derived from the mean and standard deviation of monthly precipitation (Table 2.2) and the mean and standard deviation of monthly air temperature (Table 2.3) determined for the three locations. Meteorological data were obtained from the U.S. Historic Climatology Network, from

spatial interpolation of these data (W. W. Hargrove, ORNL), or from the Hubbard Brook gopher server. The meteorological records for years 1968-1993 from Buckley, WA near the Pack Forest were used to obtain monthly temperature and precipitation means and standard deviations for the Douglas-fir simulations. These meteorological records were checked against non-continuous records obtained from Richard Hinshaw at Pack Forest and mean values were similar. Meteorological data from Athens, Georgia for years 1965-1994, supplemented with standard deviation values from Covington, GA, were used for the Loblolly pine modeling. Precipitation data from watershed 6 and temperature data from weather station 1 at Hubbard Brook Experimental Watershed in New Hampshire for years 1965-1994 were used in simulations of the eastern deciduous forest.

Table 2.1 Input parameters for three forest ecosystems.

Parameter	Douglas-fir Pack Forest, WA	Loblolly Pine Athens, GA	Deciduous Forest Hubbard Brook, NH
Latitude (¡N)	46.8	34.0	43.9
Longitude (¡W)	122.3	83.3	71.8
Begin season (day)	91	87	135
End season (day)	305	312	288
Growing season (day)	214	225	153
Field capacity (mm)	380	270	320
Wilting point (mm)	170	60	180
Available soil water (mm)	210	210	140
Length of simulation (y)	100	50	500

Table 2.2 Monthly mean and standard deviation values for precipitation (mm) used at three forest ecosystem sites in LINKAGES simulations.

	Douglas-fir		Loblolly Pine		Deciduous Forest	
Month	Mean	SD	Mean	SD	Mean	SD
January	154	64	132	53	106	56
February	119	60	114	51	96	51
March	116	38	144	66	117	39
April	109	39	98	46	106	27
May	84	34	116	56	121	66
June	74	38	99	50	126	66
July	39	32	121	68	111	46
August	51	47	110	55	136	60
September	71	48	95	48	119	46
October	92	49	87	54	116	48
November	168	69	103	54	133	33
December	151	62	115	51	129	55
TOTAL	1228		1334		1416	

Table 2.3 Monthly mean and standard deviation temperature (°C) values used at three forest ecosystem sites in LINKAGES simulations.

	Douglas-fir		Loblolly Pine		Deciduous Forest	
Month	Mean	SD	Mean	SD	Mean	SD
January	3.2	2.0	5.3	2.7	-8.5	2.0
February	5.4	1.8	7.2	2.1	-7.0	2.4
March	7.0	1.3	11.5	1.7	-1.9	1.7
April	9.1	1.2	15.8	1.2	4.4	1.7
May	12.3	1.0	19.8	1.2	11.4	1.6
June	15.1	1.3	23.8	1.1	16.1	1.4
July	17.2	1.1	25.8	1.2	18.7	1.4
August	17.4	1.2	25.2	1.1	17.7	1.3
September	14.7	1.3	22.1	1.2	13.1	1.3
October	10.2	1.0	16.2	1.6	6.9	2.2
November	5.9	1.7	11.2	1.9	1.0	1.7
December	3.3	1.9	6.9	2.2	-5.5	2.5
Annual Mean	10.1		15.9		5.5	

The attributes of tree form, longevity, and other species-specific variables required for LINKAGES simulations were obtained from literature and personal communication with scientists knowledgeable about the species of interest. Some model variables were adjusted to give output results that calibrated with field data or with, in the case of loblolly pine, another simulation model. Dr. Darrell C. West (Environmental Sciences Div., Oak Ridge National Lab.) provided considerable support with the identification and communication with scientists having specific knowledge of the three forest ecosystems. Aspects of the LINKAGES model applications to the three forest sites are outlined below.

2.2.1 Douglas-fir Plantation

The input parameters were adjusted to approach a reasonable height and basal area at 100 years which is the plantation rotation length selected for simulation. A maximum tree diameter of 2.6 m was set to that of a large tree but not the largest tree of record. Maximum tree height of 100 m was obtained from the silvics manual (Burns, 1990). The yield tables of Curtis et al. (1982) started with a site index (at 50 years) of 26 m; however, a site index of 32 m was used for calibration of basal area. The basal area from the yield table at 50 years is 45.28 m^2/ha and at 100 years is 69.5 m^2/ha. LINKAGES calculates basal areas of 47.9 m^2/ha at 50 years and 68.2 m^2/ha at 100 years after these calibration adjustment, showing close agreement with the published yield table.

The stand was initially loaded with 988 stems/ha, and no additional seedlings were allowed to germinate. No commercial thinning was applied in the simulations. Yield tables (Curtis et al., 1982) provided mortality information based on site index and number of stems planted for stands without commercial thinning. For a site index of 32 m at 50 years and 988 stems/ha, the mortality at 100 years is projected to be 578 stems/ha. A probability function in LINKAGES implements this rate of mortality.

Personal communications were made with the following:

Dan Binkley - Dept. of Forestry, Colorado State Univ., Ft. Collins, CO
Mark E. Harmon - USDA. Forest Service, Corvallis, OR
Dale W. Johnson - Desert Research Institute, Reno, NV
W. M. Post - Environmental Sciences Div., Oak Ridge National Lab., Oak Ridge, TN
Allen M. Solomon - U.S. EPA, Corvallis, OR

2.2.2 Loblolly Pine Plantation

The PTAEDA2 model of Burkhart et al. (1987) was used to calibrate LINKAGES. PTAEDA2 was run for a site index (at 25 years) of 18.3 m and 21.3 m. For each run, 988 stems/ha were planted on a 3 x 3 m grid and no thinning was applied. Basal area was used for calibration. At year 50 for a site index of 18.3 m, PTAEDA2 calculated a basal area of 28.93 m^2/ha and for a site index of 21.3 m, the basal area was 23.97 m^2/ha. At 50 years, LINKAGES calculated a basal area of 24.9 m^2/ha. Input values for height (49.7 m) and diameter (1.8 m) were obtained from the 1990 silvics manual (Burns, 1990). The calibrated model was run for 50 years in the biosolids simulations for loblolly pine.

Mortality is based on personal communication with Dr. Klaus Steinbeck at The Univ. of GA. At a planting density of 988 stems/ha on a 3 x 3 m grid there would be 10% mortality in the first 3 years, and after 50 years 618 stems/ha would remain. A probability function in LINKAGES implements this mortality rate.

Since the initial age of the planted stems in LINKAGES is assumed to be 3 years old, the stand was initially loaded with 889 stems/ha (10% mortality in the first 3 years). No additional seedlings were allowed to germinate. No thinning was applied. The planting of 988 stems/ha assumes the trees would be harvested for saw timber.

Personal communications were made with the following:
Ralph L. Amateis - Dept. Forestry, Virginia Tech., Blacksburg, VA
Klaus Steinbeck - School of Forestry, Univ. of GA, Athens, GA
Tom Waldrop - USDA. Forest Service, Clemson, SC

2.2.3 Eastern Deciduous Forest

The input parameters for most of the nine forest species considered were derived from the 1990 silvics reference (Burns, 1990). The input parameters for the less common hardwoods that were not available in the literature were obtained through personal communication. Biomass summaries from Hubbard Brook (Whittaker et al., 1974) were used to calibrate the dominant tree species of *Acer saccharum* (sugar maple), *Betula allegheniensis* (yellow birch), and *Fagus grandifolia* (beech) and good agreement was obtained. The LINKAGES model is well suited for simulation of multi-species stands. Six understory species were included in the simulations as follows: *Acer pensylvanicum, Acer spicatum, Fraxinus americana, Picea rubens, Prunus pensylvanica, and Sorbus americana.*

Personal communications were made with the following:
E. R. Buckner - Dept. of Forestry, Univ. of TN, Knoxville, TN
Christopher Eager - USDA. Northeastern Forest Experimental Station, Durham, NH

J. W. Hornbeck - USDA. Northeastern Forest Experimental Station, Durham, NH
W. B. Leak - USDA. Northeastern Forest Experimental Station, Durham, NH
Wayne Martin - USDA. Northeastern Forest Experimental Station, Hubbard Brook Exp. Watershed, NH
John Pastor - Dept. of Forestry, Univ. of MN, Duluth, MN
W. M. Post - Environmental Sciences Div., Oak Ridge National Lab., Oak Ridge, TN
T. G. Siccama - Dept. of Forestry, Yale Univ., MA

2.3 APPLICATION SCENARIOS

The three forest ecosystems of Douglas-fir on the coast range of Washington state, loblolly pine on Piedmont soil of Georgia, and eastern deciduous forest in New Hampshire were selected following recommendations from the Problem Formulation Workshop (Chapter 1).

Biosolids application rates were selected to cover a range of rates that might be applied by municipalities or in commercial applications. Simulations were conducted with 0, 5, 10, 20, and 40 Mg/ha as single biosolids applications and with repeat applications of 5 and 10 Mg/ha at three-year intervals for a total application of 35 and 70 Mg/ha (7 additions over 19 years).

Biosolids applications were made in the first year of simulation for the Douglas-fir and Loblolly pine sites corresponding with the year of stand establishment. A 5-year delay in biosolids application was made in the deciduous forest to represent application to a well-developed coppice regrowth stand which develops quickly from root systems. In all LINKAGES simulations, trees are established with a 1.43 cm stem dbh, thus the seedling stage is not considered and the trees are established at an age of about three years in the model.

2.4 SIMULATION RESULTS AND DISCUSSION

2.4.1 Northwest Douglas-fir Forest

Douglas-fir showed substantial increases in aboveground growth with increase in biosolids application (Fig 2.2) particularly with applications above 10 Mg/ha. Mean biomass values were determined from 100 replicate plots that were simulated stochastically with LINKAGES, and 95% confidence intervals were also determined from the model output. Statistically significant growth responses at years 50 and 100 were obtained with all biosolids applications (Table 2.4). The low confidence interval values are due to the large number of replicates used in the simulations.

Net primary production (NPP) is the annual gain in carbon fixation minus respiratory and mortality losses for the year, thus, NPP shows the net annual growth increment. Mean NPP values determined from simulations of 100 plots along with 95% confidence intervals (Table 2.5) show that Douglas-fir has increasing NPP with increase in biosolids application rate. NPP was also much higher at 100 years than at 50 years. The multiple application rates gave greater NPP values than any single application rate. The seven applications of 5 Mg/ha given over 19 years (7x5) had a lower total biosolids application than the 40 Mg/ha treatment but had more than 20% greater NPP at years 50 and 100. The associated enhanced aboveground biomass (Table 2.4) may or may not be sufficient to cover the cost of multiple applications of biosolids.

Table 2.4. Mean and 95% confidence interval for aboveground biomass of Douglas-fir simulated at 50 and 100 years in response to biosolids applications.

	Year 50		Year 100	
Application (Mg/ha)	Mean (Mg/ha)	95%CI	Mean (Mg/ha)	95%CI
0	292	4.3	482	14.1
5	315	4.6	525	15.5
10	338	5.0	569	16.8
20	382	5.6	656	19.5
40	464	7.0	823	24.8
7 X 5[a]	540	8.1	984	29.8
7 X 10	783	12.0	1522	47.1

[a] Seven applications @ 5 Mg/ha each

Table 2.5. Mean and 95% confidence interval of aboveground net primary production for Douglas-fir simulated at 50 and 100 years in response to biosolids applications.

	Year 50		Year 100	
Application (Mg/ha)	Mean (Mg ha^{-1} y^{-1})	95%CI	Mean (Mg ha^{-1} y^{-1})	95%CI
0	6.4	0.1	9.1	0.4
5	6.9	0.1	10.0	0.4
10	7.5	0.1	11.0	0.4
20	8.6	0.2	13.0	0.5
40	10.9	0.2	16.9	0.7
7 X 5[a]	13.3	0.2	20.6	0.8
7 X 10	20.8	0.4	33.6	1.4

[a] Seven applications @ 5 Mg/ha each

Soil organic matter is enhanced by biosolids applications relative to the control treatment (Fig. 2.3). A decline in organic matter is projected during the first 50 years of plantation growth, but this changes to an increase as mortality increases with plantation age.

2.4.2 Southeastern Loblolly Pine Plantation

Greater growth of Loblolly pine was obtained with biosolids applications particularly after 15 years of plantation growth (Fig. 2.4). The pattern of results is similar to those for Douglas-fir, however, aboveground biomass at 50 years is about half of that for Douglas-fir. A comparison of loblolly pine biomass values at 25 and 50 years shows statistically significant increases in biomass in most cases with increase in biosolids application (Table 2.6). The 95% confidence intervals were determined from 100 replicate plot simulations. The multiple applications were superior to single applications as shown by the greater growth with the 7 x 5 Mg/ha application than the 40 Mg/ha application. Net primary production (Table 2.7) ranged by more than a factor of 2 with the biosolids treatments.

Soil organic matter increased with biosolids applications (Fig. 2.5) and the pattern of results were similar to those obtained for Douglas-fir (Fig. 2.3). The multiple biosolids

applications of 5 and 10 Mg/ha had higher soil organic matter levels after 15 years than those obtained with a single application.

Table 2.6. Mean and 95% confidence interval of aboveground biomass for Loblolly pine simulated at 25 and 50 years in response to biosolids applications.

	Year 25		Year 50	
Application (Mg/ha)	Mean (Mg/ha)	95%CI	Mean (Mg/ha)	95%CI
0	69	1.7	132	3.2
5	72	1.8	140	3.4
10	75	1.9	148	3.6
20	80	2.0	163	3.9
40	87	2.3	190	4.6
7 X 5[a]	95	2.5	230	5.5
7 X 10	102	2.7	300	7.3

[a] Seven applications @ 5 Mg/ha each

Table 2.7. Mean and 95% confidence interval of aboveground net primary production for Loblolly pine simulated at 25 and 50 years in response to biosolids applications.

	Year 25		Year 50	
Application (Mg/ha)	Mean (Mg ha^{-1} y^{-1})	95%CI	Mean (Mg ha^{-1} y^{-1})	95%CI
0	4.4	0.1	2.5	0.1
5	4.7	0.1	2.6	0.1
10	5.0	0.1	2.8	0.1
20	5.6	0.2	3.1	0.1
40	6.6	0.2	3.8	0.2
7 X 5[a]	8.2	0.2	4.9	0.2
7 X 10	10.0	0.4	7.1	0.3

[a] Seven applications @ 5 Mg/ha each

2.4.3. Eastern Deciduous Forest

Simulations for this natural forest ecosystem were conducted over a much longer time span (500 y) than for the conifer plantations. Due to species diversity in the deciduous forest, 500 replicate plots were simulated for estimation of mean output values and 95% confidence intervals. Growth responses to biosolids application were apparent by the third decade, however, these effects dissipated after year 350 (Fig. 2.6). The greatest difference between the applications occurred at about year 200 with a 14.8% increase in aboveground biomass for the 7x10 Mg/ha treatment relative to the control. The response of the deciduous forest to biosolids was relatively much less than that of the conifer plantations. Aboveground biomass at years 50 and 100 showed several significant responses to biosolids applications (Table 2.8). In contrast with the conifer results, the deciduous forest did not show preferential response to the multiple applications. The aboveground biomass for the 7x5 treatment was not statistically different from the 20 Mg/ha application. Similarly, the 7x10 treatment had the same biomass as the 40 Mg/ha application (Table 2.8).

Table 2.8. Mean and 95% confidence interval of aboveground biomass for eastern deciduous forest simulated at 50 and 100 years in response to biosolids applications.

	Year 50		Year 100	
Application (Mg/ha)	Mean (Mg/ha)	95%CI	Mean (Mg/ha)	95%CI
0	91	1.1	232	4.1
5	96	1.2	243	4.3
10	99	1.2	249	4.4
20	104	1.4	259	4.9
40	111	1.6	275	5.4
7 X 5[a]	105	1.4	262	5.0
7 X 10	111	1.6	275	5.4

[a] Seven applications @ 5 Mg/ha each

Small differences in net primary production were simulated at years 50 and 100 for the various biosolids applications (Table 2.9). The largest differences in NPP values were 0.7 and 1.0 Mg ha^{-1} y^{-1} for years 25 and 50 respectively. These are relatively small effects compared to those obtained for the conifer simulations. The available water capacity of the shallow soils at Hubbard Brook was 67% of those for the conifer sites (Table 2.1), and soil water deficit was more likely to become a limiting factor at Hubbard Brook than at the other two sites. Water deficit may have limited tree growth response to biosolids in addition to the relatively short growing season at Hubbard Brook (Table 2.1).

Table 2.9. Mean and 95% confidence interval of aboveground net primary production for eastern deciduous forest simulated at 50 and 100 years in response to biosolids applications.

	Year 50		Year 100	
Application (Mg/ha)	Mean (Mg ha^{-1} y^{-1})	95%CI	Mean (Mg ha^{-1} y^{-1})	95%CI
0	7.5	0.1	8.5	0.2
5	7.7	0.1	8.7	0.2
10	7.8	0.1	8.8	0.2
20	7.9	0.1	9.0	0.2
40	8.2	0.1	9.5	0.2
7 X 5[a]	8.0	0.1	9.2	0.2
7 X 10	8.2	0.1	9.5	0.2

[a] Seven applications @ 5 Mg/ha each

The soil organic matter of the deciduous forest increased with biosolids application (Fig. 2.7); however, the pattern of the results contrasted with those of the conifer simulations (Figs. 2.3, 2.5). Rapid accumulation of soil organic matter occurred after an initial decline during the first two decades. The effect of biosolids on soil organic matter dissipated after 400 years. The accumulation of organic matter at the Hubbard Brook site is due in part to the low annual mean temperature which limits decomposition of litter components.

The most significant change in the forest community to the application of biosolids is the enhanced growth of *Betula allegheniensis* (yellow birch) with high application rates. Yellow birch is a species that responds readily to increases in available nitrogen. The biomass of the other two dominant species, *Acer saccharum* (sugar maple) and *Fagus grandifolia* (beech) were largely unaffected by the biosolids applications (Fig. 2.8). Six other species were in the forest simulations but they had negligible contributions to stand biomass. Nevertheless, there was some decline in the minor species with the biosolids treatments that can be attributed to the enhanced growth of yellow birch. In particular, *Picea rubens* (red spruce) biomass was 36% lower with 40 Mg/ha of biosolids at year 100 than in the control plot. Selected simulations in Fig. 2.9 show a decline in aboveground biomass of red spruce with increasing application of biosolids. A small increase in *Prunus pensylvanica* (pin cherry) occurred during the first 50 years with higher applications of biosolids, and *Sorbus americana* (mountain-ash) followed a similar pattern over a time span of 100 years (Fig. 2.9).

2.5 UNCERTAINTY ANALYSIS

Biosolids have considerable variation in nutrient and contaminant content due to a range of available waste sources and treatment procedures. The influence of this variability on simulation was investigated. In addition, vegetation and soil attributes of a plantation or forest have natural variation. Uncertainty analysis was conducted to determine if variable growth responses from variability in biosolids composition, vegetation, and soil properties could be statistically distinguished between the biosolids applications and the control plot simulations.

The Latin hypercube sampling method was used to propagate frequency distributions (i.e., range of variability or uncertainty) of selected input variables through the LINKAGES model. This sampling method was accomplished by combining the PRISM code of Gardner et al. (1983) with LINKAGES. In the procedure, the input distributions of variables with uncertainty are subdivided into 200 equal probability classes. Next, 200 sets of input values are generated with PRISM by sampling each of the input distributions without replacement, and LINKAGES was executed 200 times with these 200 input datasets for each of the seven biosolids application scenarios and each of the three forest types. The output from these model runs generated frequency distributions of outputs and these are summarized for the three forest ecosystems.

The determination of frequency distributions of model inputs was based on literature or by assuming some variability. A survey of chemical content of biosolids showed the Kjeldahl N content to have a normal distribution with a mean of 45.90 mg/g and a standard deviation of 26.95 mg/g. These values were determined from recalculation of statistics obtained from the National Sewage Sludge Survey (Table 2.10). Similarly the nitrite and nitrate content of biosolids is variable and this was represented as 0.65% of the Kjeldahl N values, giving the mineral N values a normal distribution. This proportion of mineral N in biosolids was calculated from data in the National Sewage Sludge Survey.

The uncertainty analysis included some soil and plant properties of the three forest applications. Mean values for field capacity and wilting point of the soils (Table 2.1) were obtained from the STATSGO database (U.S. Department of Agriculture, Washington, DC). A standard deviation of 10% for these means was chosen as a modest estimate of soil variability represented by a normal distribution and minimum and maximum values of ± 3 standard

deviations. This range of soil water availability resulted in some low values of available soil water that induce water limitations on tree growth.

Table 2.10. Summary statistics for concentrations of forms of nitrogen in sewage sludge from the 1988 sewage sludge survey (dry weight).

Analyte	N	detects	maximum (mg/kg)	mean (mg/kg)	standard deviation	95% UCL[1]
nitrate	177	169	35322	963.89	3630	1415
nitrite	177	158	17700	162.62	1895	598.08
total Kjeldahl N	178	100	129635	45902	26949	49251

[1] UCL, upper confidence limit around the mean

The mean values of five nitrogen parameters used in LINKAGES decomposition and growth algorithms were given normal distributions with standard deviations of 3% of the mean and minimum and maximum values that were ± 3 standard deviations. These choices provided modest variability of nitrogen dynamics in the ecosystems. The major variability examined in the uncertainty analysis was due to the nitrogen content of biosolids which had a coefficient of variation of 59%. Examples of uncertainty effects on forests treated with biosolids are shown in simulations from the three ecosystems.

2.5.1 Douglas-fir Forest

The frequency distribution results simulated for the aboveground biomass of Douglas-fir were normally distributed for each of the seven biosolids treatments. Example results for the 5, 40, and 7x5 Mg/ha treatments obtained at 50 years (Fig. 2.10) and at 100 years (Fig. 2.11) showed statistically significant differences with the multiple application giving greater biomass than single applications. The results for the 7x10 Mg/ha application were the highest of all the treatment comparisons. Results obtained for basal area of forest, leaf litter mass, and soil organic matter followed the pattern of results obtained for aboveground biomass.

2.5.2 Loblolly Pine Plantation

The aboveground biomass frequency distributions for loblolly pine showed similar results to those for Douglas-fir. The response of soil organic matter to biosolids treatments also showed a similar pattern of responses. A comparison of 5, 40, and 7x5 Mg/ha application rate effects on soil organic matter (Fig. 2.12) showed significant differences in organic matter with these treatments. The multiple application treatment enhanced soil organic matter more than the single application treatments.

2.5.3 Eastern Deciduous Forest

The output frequency distributions for the deciduous forest were generally skewed to the right as shown in the comparisons of leaf litter for the 5, 40, and 7x5 Mg/ha applications at year 100 (Fig. 2.13). The skewed distributions were statistically compared with a non-parametric test procedure available in SAS called NPAR1WAY. In this test, similarly skewed distributions can be compared for significant lateral shifts between any of the distributions. This test showed that

there were significant differences between the distributions; however, the test does not distinguish among the individual biosolids applications.

2.5.4 Uncertainties Associated with the Use of LINKAGES

- **Long-term estimates of production.** Biomass estimates for decades or centuries in the future have not been validated.

- **Parameterization of LINKAGES.** Seedling data were used to parameterize the Linkages model. The validity of these data for older trees is unknown.

- **LINKAGES Algorithms.** The prediction of organic matter decomposition by LINKAGES is somewhat uncertain because the organic matter decomposition and litter algorithms of LINKAGES have not been written to account for sewage sludge.

2.6 FIELD STUDIES OF SOUTHWESTERN SEMIARID RANGELAND

As in the Douglas-fir forest, experimental applications of sewage sludge and measurements of effects have been substantial in the semi-arid rangeland.

2.6.1 Rio Puerco Valley, NM

Municipal sewage sludge from Albuquerque, NM was applied to plots in the Rio Puerco Valley in New Mexico in 1985 by researchers with the Rocky Mountain Forest and Range Experiment Station of the U. S. Forest Service. The amendment was a single application at 22.5, 45, and 90 Mg/ha.

One year after application of sewage sludge, there was a linear increase in total soil nitrogen, phosphorus, water soluble potassium, and electrical conductivity with application rate (Fresquez et al. 1990a). Within the first two growing seasons, organic matter increased slightly in the 22.5 and 45 Mg/ha biosolids treatments, but after four growing seasons, organic matter increased linearly with application rate, and the 45 and 90 Mg/ha plots contained 18 g/kg and 24 g/kg organic matter in soil, compared to 13 g/kg for the control treatment (Fresquez and Dennis 1990).

Total nitrogen, phosphorus, potassium, and crude protein increased linearly with sludge application in blue grama, galleta, and bottlebrush squirreltail (Fresquez et al. 1990a). Deficiencies of nitrogen, phosphorus, and potassium were previously observed in New Mexico rangeland forage, but were no longer observed following sludge application (Fresquez et al. 1990a, Fresquez et al. 1991). Aluminum and iron concentrations in these plants generally decreased, with the decrease of the iron associated with lower tissue fiber in these plants (Fresquez et al. 1990a, Fresquez et al. 1991, Dennis et al. 1988).

Total plant density, species richness, and species diversity decreased with sludge application (Fresquez et al. 1990a). Differences in plant density (plants per $0.5m^2$) became statistically significant at the 0.05 level (TukeyÕs multiple range test) in the third and fourth growing seasons following application (Fresquez et al. 1990b). During the first growing season following sewage sludge application, broom snakeweed, bottlebrush squirreltail, ring muhly and fendler three awn decreased in density with increasing rates of application (Fresquez et al.

1990b). Broom snakeweed was one of three dominant plant species in the plots prior to application. The snakeweed plants that survived increased in cover and biomass. The number of plant species decreased from 16 to 10 species in the 90 Mg/ha treatment. These effects continued through the 4th growing season following application (Fresquez et al. 1990b).

The total herbaceous plant biomass increased with sewage sludge amendment, especially with the addition of 45 Mg/ha (Fresquez et al. 1990a). At sludge applications ranging from 22.5 to 90 Mg/ha, blue grama production ranged from 1.5 to 3 times production in control plots. The increased production remained for at least 5 years, though it had substantially decreased for the lowest (22.5 Mg/ha) application (Fresquez et al. 1991). Plant yields at 90 Mg/ha were no higher than those at lower application rates (Fresquez et al. 1990a). Broom snakeweed, a toxic, inedible range plant, decreased with sludge application over four years following treatment (Fresquez 1990b).

Microbial populations including ammonium oxidizers, other bacteria, and fungi increased linearly with increasing sewage sludge amendments (Dennis and Fresquez 1989); however, the diversity of fungi decreased (Fresquez and Dennis 1990). Sludge applications of 22.5, 45, and 90 Mg/ha decreased the numbers of fungal species to 38%, 30% and 11% of the fungi observed in the control soils. *Mucor* spp. and *Penicillium chrysogenum* were dominant in the first growing season following application, and *Penicillium janthinellum* were dominant after the second season. The former two species are often found in sewage sludges (Fresquez and Dennis 1990). Fungal diversity returned to control levels after the second growing season in the 22.5 and 45 Mg/ha treatments. In the 22.5 Mg/ha plot, all fungal parameters returned to control values after three growing seasons. However, after four years, the higher sludge applications were still associated with higher fungal propagules than the control (Fresquez and Dennis 1990).

2.6.2 Sevilleta National Wildlife Refuge, NM

Municipal sewage sludge from Albuquerque, NM was applied to plots in the Sevilleta National Wildlife Refuge in New Mexico in 1991 by researchers with the Rocky Mountain Forest and Range Experiment Station of the U. S. Forest Service. The amendment was a single application at 45 Mg/ha. In general, effects of sludge were tied closely to precipitation.

Municipal sewage sludge applied at 45 Mg/ha increased the total Kjeldahl nitrogen and extractable ammonium, nitrate, and phosphorus in surface soil and total nitrogen in plants, but did not increase soil organic carbon in the soil below the sludge in the two years following application (Aguilar et al. 1994b). The researchers speculated that the organic matter of the soil would never increase since sludge was not incorporated. The C:N ratio of the sewage sludge was lower than that of the soil on the plots (Aguilar et al. 1994b).

The plant cover increased after 1.5 years on sludge-amended plots which received substantial rainfall (50 mm in addition to the natural quantity) in a precipitation simulation study (Aguilar et al. 1994b). The response was mostly from cool season forbs and shrubs rather than warm season grasses (Loftin and Aguilar 1994). The following fall was dry, and plant cover decreased to a greater degree in sludge treatments than in control plots (Loftin and Aguilar 1994). Plant root growth was much lower on plots treated with 45 Mg/ha of sludge than on untreated plots.

Sludge applied at 45 Mg/ha increased the decomposition of blue grama litter, both above- and below-ground, in the second but not the first year following application. On the other hand, sludge amendment did not affect the decomposition of saltbush litter (Aguilar et al. 1994b). The observed difference may have been due to the different C:N ratios for the plants, 167:1 for the blue grama and 31:1 for the saltbush (Aguilar et al. 1994b) or to the higher percentage of water soluble constituents in the saltbush (Loftin and Aguilar 1994).

2.6.3 Meadow Springs Ranch, Larimer County, CO

Municipal sewage sludge from Fort Collins, CO was applied to plots in northern Larimer County, CO in 1991 by researchers at Colorado State University. The amendment was a single application at 2.5, 5, 10, 21, and 30 Mg/ha. In the two years following application, ammonium and nitrate nitrogen increased in soil at depths of up to 30 cm (Harris-Pierce 1994).

Total canopy cover, including perennial forbs and grasses, increased following the application of sewage sludge (Harris-Pierce 1994). The cover was no greater for 30 Mg/ha, the highest application rate, than for 21 Mg/ha. Although total biomass did not increase with any application, the biomass of perennial forbs and grasses, especially warm season grasses, increased in years 1 and 2. The increase in forbs was expected with the addition of nitrogen, though Harris-Pierce (1994) cites studies that show little effect of nitrogen on community composition. Shrub biomass, particularly fringed sage, increased in the first year. All four dominant species took up nitrogen, phosphorus and potassium to higher tissue levels than in the absence of sludge.

2.6.4 Wolcott, CO

Municipal sewage sludge from the Upper Eagle Valley Consolidated Sanitation District, Colorado, was applied to plots in Wolcott, CO in 1991 by researchers at Colorado State University. The amendment was a single application at 5, 10, 15, 20, 25, 30, 35, and 40 Mg/ha.

No significant changes in plant community composition were observed in this sagebrush ecosystem. It is possible that there was an increase in species diversity of perennial forbs (Pierce 1994).

Canopy cover did not significantly increase in the two years following biosolids application, but mean values at the 20 to 30 Mg/ha rates were the highest (Pierce 1994). Intermediate treatment rates had some increase in canopy cover of annuals in the two years following application-- from 0 to 3% at intermediate treatment rates. Pierce (1994) states that species diversity of perennial forbs increased, but the particular treatments that resulted in the effect were not identified.

No significant changes in plant biomass were observed with increasing application rate, nor were there shifts in biomass of annuals vs perennials, lifeform or seral stage. Nitrogen and phosphorus uptake was increased in perennial grasses, possibly providing increased forage quality, but not increased biomass.

2.6.5 Sierra Blanca Ranch, Texas

Municipal sewage sludge from New York City has been applied annually to plots in the Chihuahuan Desert since 1992 by researchers at Texas Tech University and MERCO Joint Venture. Application rates are about 0, 7, 18, 34, and 90 Mg/ha.

The aboveground biomass of tobosagrass increased with three annual sludge applications of about 7, 18, and 34 Mg/ha, but not 90 Mg/ha relative to the control (Wester and Benton 1995). Alkali sacaton biomass was increased at 18 and 34 Mg/ha, but not at higher or lower application rates. An explanation of the failure of the 90 Mg/ha rate to increase biomass is that in a dry growing season, precipitation is absorbed by biosolids.

Researchers have begun to study microbial degradation of biosolids, but results are not available yet.

2.6.6 Utah

Sabey et al. (1990) studied the growth of fourwing saltbush and western sagebrush following sewage sludge application. Because the studies were conducted with copper mine spoil, in pots rather than in the field, and with biosolids mixed into the soil, the growth responses are not detailed here. However, this is the only study in which the effects of sewage sludge on mycorrhizal associations with plants were studied.

Sagebrush is generally associated with mycorrhiza. Sewage sludge was added to pots at rates equivalent to 0, 30, and 60 Mg/ha, and sagebrush roots were surrounded with sufficient soil to serve as an inoculum. Colonization of plant roots was 31.8% for 30 Mg/ha applications, 18.4% for controls, and 3.5% for 60 Mg/ha applications (Sabey et al.1990).

2.6.7 Conclusions

Nutrients in municipal sewage sludge applied to semi-arid rangeland may have numerous effects. Organic matter in soils, forage nutrients, total plant biomass, canopy cover and microbial biomass were observed to increase with sludge addition. Many effects were dependent on precipitation. Broom snakeweed, a toxic, inedible range plant, decreased in biomass following the application of biosolids, though the study was not duplicated. On the other hand, plant density, species diversity, and species richness, as well as fungal diversity decreased with the addition of sludge, though the duration of these effects is not known. The colonization of sagebrush roots by mycorrhizae was also adversely affected by the addition of high levels of municipal sewage sludge. Except for the Sierra Blanca Ranch, all research sites have received single applications of sewage sludge. Long-term effects of multiple applications are being studied at the Sierra Blanca Ranch, Texas. There are indications that plant response depends on season of application, number of years of application, growing conditions during the year of application, and growing conditions following application.

3. POTENTIAL FOR LEACHING AND EROSION

A significant concern with application of biosolids to natural ecosystems is pollution of surface waters and groundwater by erosion and leaching. Some evaluations of these issues have been made and these are summarized from the literature. Some inferences are also drawn from the water budget simulation results.

3.1 NORTHWESTERN DOUGLAS-FIR FOREST

The water budget for this site shows a mean annual precipitation of 1228 mm and this was associated with a mean annual evapotranspiration of 595 mm in LINKAGES simulations. The difference between precipitation and evapotranspiration is the net drainage for the site since on an annual basis change in soil water storage is essentially zero. The net drainage is estimated to be 633 mm/y and this is 52% of the annual precipitation. There is significant seasonality in precipitation in the Pacific Northwest with less precipitation during the May to October period (Table 2.3), and this, combined with high evapotranspiration, greatly reduces the erosion and leaching potential during summer and early autumn. Conversely, higher precipitation during November to April increases the potential for leaching and erosion during this period, particularly after soil water has been recharged from the evapotranspiration losses. Nitrate leaching from biosolids requires the combination of mineralization of organic nitrogen as well as soil water movement. Mineralization can proceed even under snow pack conditions and nitrate leaching can be expected under conditions that produce soil water drainage to groundwater.

Exceedingly high rates of biosolids application can lead to groundwater contamination. Riekerk (1978) showed maximum nitrate concentrations of about 120 mg/L in groundwater of a 45-year Douglas-fir stand treated with biosolids at an application rate of 4.8 Mg N/ha. The nitrate levels in groundwater remained elevated for several years after this application. Further investigations of Riekerk (1981, 1982) suggest that nitrogen applications up to 500 kg N/ha in Douglas-fir plantations do not lead to significant nitrate leaching losses. Biosolids applications above 400 kg N/ha do not provides additional growth response to the additional nitrogen in Douglas-fir stands (Cole et al. 1983), thus applications above this level should be avoided unless carefully justified. In the simulations (Chapter 2), application rates of 5, 10, 20, 40, 7x5, 7x10 Mg/ha correspond to nitrogen loadings of 230, 459, 918, 1840, 1610, and 3210 kg/ha respectively. Thus significant nitrate leaching would not be expected from the 5 or 10 Mg/ha rates of biosolids application.

3.2 SOUTHEASTERN LOBLOLLY PINE PLANTATION

The water budget for the loblolly pine plantation has a mean annual precipitation of 1334 mm and this was associated with a mean annual evapotranspiration of 802 mm in LINKAGES simulations. Net drainage is estimated to be 532 mm/y and this is 40% of the annual precipitation. Precipitation is somewhat uniform through the year, however, the loss of soil water during the growing season by evapotranspiration greatly reduces the potential for leaching and erosion during the May through October period.

A comprehensive study of one-time liquid- and solid-biosolids applications to loblolly pine plantations was undertaken on the Savannah River Plant of the U.S. Department of Energy. A compilation of results from the studies (Davis 1989) showed increased forest productivity with biosolids applications up to 400 kg N/ha without any environmental degradation of the sandy

coastal plain soils. Nitrate-N levels in soil solution, sampled with fritted glass lysimeters in the plots, were up to 70 mg/L at 1 m soil depth in the highest liquid-biosolids application of 800 kg N/ha during the first year. Nitrate levels were much higher in soil solution from liquid than solid applications, and levels were higher in 1-year and 28-year plantations than in the 3- and 9-year plantations investigated. At 3 years after application, soil nitrate levels were close to control plot values of about 1 mg N/L. Groundwater wells, monitored for six years in the vicinity of the experimental plots, did not show any elevated nitrate concentration due to biosolids application. Groundwater was between 9 and 28 m depth at the various sites used for the research. The investigators recommended that forest areas along drainage ways not receive biosolids to prevent contamination of surface waters. A strip 100 m wide was recommended for riparian zones. Other investigations have shown that riparian forest strips can effectively reduce nitrate levels in runoff from upslope farm-animal lands through denitrification and plant uptake (Hubbard and Sheridan 1995). The same benefit can be realized with riparian buffer strips around upslope biosolids application areas. A strip 100 m wide may be overly conservative for protection of surface water quality. Cooper et al. (1986) found that riparian strips as narrow as 16 m were effective in removal of nitrate from cultivated fields. Watershed management of biosolids application can reduce the hazard of nitrate leaching by inclusion of untreated riparian strips along waterways and the use of a staggered biosolids application so that no more than $^1/_3$ of the watershed is treated in a given year. Nitrate levels on application areas can be reduced to background levels by the third year. This ecological risk assessment is focused on plantations on Piedmont soil areas, and the leaching results from investigations on coastal plain soils at the Savannah River Site may not be strictly applicable. The clay subsoils of the Piedmont tend to favor lateral subsurface flow which discharge to surface water in low positions in the landscape. Thus, biosolids application on Piedmont soils may lead to eutrophication of surface waters more than groundwater contamination.

3.3 EASTERN DECIDUOUS FOREST

The water budget for the deciduous forest has a mean annual precipitation of 1416 mm, and this was associated with a mean annual evapotranspiration of 538 mm in LINKAGES simulations. Net drainage is estimated to be 878 mm/y, and this is 62% of the annual precipitation. Evapotranspiration simulated with LINKAGES is higher than the value of 423 mm/y used in simulations for Hubbard Brook by Crohn and Haith (1994). Nevertheless, a high potential for leaching losses of chemicals from biosolids exists at the Hubbard Brook site due to the large excess of precipitation over evapotranspiration during winter and early spring. Another feature at Hubbard Brook that favors leaching is the relatively shallow soil depth over bedrock. Hillslopes with shallow soils generate lateral subsurface flow in response to precipitation. Forests on such soils are vulnerable to leaching of soluble chemicals from biosolids application. Even deeply weathered forest soils, such as at Walker Branch watershed in eastern Tennessee, generate shallow subsurface flow in response to precipitation (Luxmoore et al. 1990). In this case, clay subsoils restrict vertical drainage causing shallow subsurface flow on hillslopes during storm events. A significant proportion of eastern deciduous forest occurs on hilly terrain in the eastern United States. Thus, leaching is an important consideration in biosolids applications.

Modeling analyses of nitrate leaching from sewage sludge applications in a northern hardwood forest (Hubbard Brook) were reported by Crohn and Haith (1994) and Crohn (1995b). They used a forest succession model called FORSENTO to estimate the amount of biosolids that may be safely applied to a deciduous forest without causing nitrate-N concentration to exceed 10 mg/L in groundwater, the drinking water standard. The FORSENTO model has many similarities

to the LINKAGES model used in the present investigation. Algorithms for transformation of biosolids in soil resulted in increases in inorganic nitrogen in FORSENTO which increased tree growth. Application of 4.5 Mg/ha of anaerobically digested biosolids (225 kg N/ha) at 3-year intervals to a 31-year-old deciduous forest was predicted to have nitrate leachate concentrations less than the drinking water standard 99% of the time. These simulations were conducted with annual time-step modeling which is a significant limitation to representation of precipitation-driven leaching processes that occur at much finer time scales. An annual average nitrate concentration less than the drinking water standard could involve several leaching events, particularly during late winter and early spring, that far exceed the drinking water standard. Field investigations and modeling of solute transport in soils are active areas of research.

An early biosolids field study, conducted at Hubbard Brook Experimental forest by Koterba et al. (1979), did not include soil water sampling during the December through May period when their lysimeters were frozen. Thus, the effect of snow melt was not evaluated. Composted biosolids were applied at 5.8 and 28 Mg/ha in this study and nitrate levels in soil water in both treatments were very low, indicative of spring and summer conditions with rapid utilization and immobilization of nitrate within the forest ecosystem. Tracy (1988) applied anaerobically digested biosolids to a mixed hardwood forest in Pennsylvania at a rate of 388 kg N/ha and found low nitrate N levels, unchanged from control levels, in soil water until mid December. At this time the soil water nitrate-N increased to about 4 mg/L. Further measurements were not made. Nitrate levels may have increased through the winter period in this study.

Crohn and Haith (1994) summarize a number of hardwood forest investigations of biosolids effects on leaching of nitrogen. Aerobically digested biosolids were applied at 200 kg N/ha in a mixed hardwood in Maryland in studies conducted by Aschmann et al. (1990, 1992). In the year following application, nitrate-N concentrations ranged from 30 mg/L in December to less than 1 mg/L in late summer. It is expected that there would be considerable seasonality in nitrate concentration in soil water associated with different rates of decomposition, uptake, and volatilization of N during the year. Greatest leaching probably occurs during snow melt in northern climates and during late winter and before leaf out in early spring for southern areas without winter snow accumulation. Soil water status is high during this period and any rainfall received readily leads to soil water drainage (leaching to groundwater) and runoff generation.

3.4 SOUTHWESTERN SEMIARID RANGELAND

3.4.1 Empirical Studies of Leaching

Leaching of nitrate and other constituents of sludge to groundwater is unlikely in the semi-arid rangeland primarily because of the depth to groundwater. At the Sevilleta National Wildlife Refuge in New Mexico, the distance to ground water is 30 m (Aguilar et al. 1994b). At the Sierra Blanca ranch in the Chihuahuan desert, Texas, the distance is 150 m (Shanks et al. 1995). Aguilar et al. (1994b) suggest that the alkaline soils and low rainfall in the southwest are impediments to the leaching of contaminants added to soil in sewage sludge. In the Rio Puerco Watershed in New Mexico, water content below one meter depth does not change, even in response to precipitation (Aguilar and Aldon 1991), which also suggests that leaching is unlikely.

Studies of sludge application at 0 to 40 Mg/ha in a semiarid sagebrush ecosystem in Wolcott, CO showed no detectable leaching of nutrients or metals beyond 90 cm for the first two years after application (Pierce 1994). The highest concentration of nitrate in lower soil horizons

was at the 20 Mg/ha rate two years after application. Harris-Pierce (1994) found increased nitrate, ammonium, and salts (electrical conductivity) in the top 30 cm of soil in Larimer County, CO, but no evidence of leaching below the root zone.

At Sevilleta Wildlife Refuge in New Mexico, the ammonium added in sewage sludge and resulting from rapid decomposition of organic compounds was found mostly in the top 5 cm and decreased after the first year following application. High levels of nitrate compared to background concentrations occurred to a depth of 80 cm and possibly deeper, though most nitrate was in the 1 to 40 cm depth range. Total Kjeldahl nitrogen increased only in the top 5 cm of soil (Aguilar et al. 1994b).

At the Sierra Blanca Ranch in Texas, calcium, magnesium, manganese, nitrate, orthophosphate, pH, sulfate, strontium, and vanadium were elevated at 30 cm below the surface in a clay loam soil. These concentrations were generally elevated in the 90 Mg/ha application rate, compared to other rates. In a sandy loam soil, about 76 cm below the surface, only barium was increased at any time after application. Chloride, potassium, and total hardness increased with sludge application in two soils (Fish et al. 1995).

3.4.2 Runoff and Erosion

Soils in arid and semi-arid rangelands erode readily because of low soil organic matter, low cover, extensive bare ground, and torrential rainfall events. Thus, the changes in erosion with application of biosolids are worthy of consideration.

Sludge applied at 45 Mg/ha to the Sevilleta National Wildlife Refuge in New Mexico decreased surface runoff (by a factor of 3.4 to 37) and total sediment yield (Aguilar et al. 1994a, Aguilar et al. 1994b). The plots had slopes of 6% and 10 to 11 %. In the first year following application, only 4 storm events produced much runoff, and the following year, no significant runoff was measured. Total suspended solids concentrations in runoff water were greater with than without sludge application, but the total volume of suspended solids was lower in treated plots. Similarly, concentrations of total Kjeldahl nitrogen, nitrate and nitrite, and total phosphorus were greater in runoff water from sludge-treated plots, but total yields of these nutrients leaving the sludge-treated plots were lower (Aguilar et al. 1994a). Most of the total Kjeldahl nitrogen was organic nitrogen.

Rainfall simulation experiments were also conducted, resulting in a factor of 27 to over 250 decrease in runoff with sludge amendment (Aguilar et al. 1994b). Runoff was not generated at rainfall inputs up to 109 mm. The reduction in runoff in sludge-treated sites was probably due to the increased roughness of ground surface as well as to the water holding capacity of the sludge itself (Aguilar et al. 1994b).

Application rates of biosolids at 20 Mg/ha and above to grasslands near Wolcott, Colorado, were associated with nitrate in surface soils (top 10 cm) above 10 mg/kg one year after application, and Pierce (1994) suggested that this level of nitrate was sufficient to cause nitrogen losses in runoff. The prediction is based on slope, soil characteristics, and exposed bare ground.

Erosion at the Sierra Blanca Ranch, Texas, was reduced considerably with the addition of 90 Mg/ha of biosolids, especially in treatments on plots with vegetation; in one soil the reduction was from 0.61 Mg/ha to 0.1 Mg/ha (Moffett and Zartman 1995). In some soils as little

as 7 Mg/ha reduced erosion (personal communication, David Wester, Texas Tech University, June 1997). Infiltration flux and cumulative infiltration increased with biosolids applications of about 7, 34, and 90 Mg/ha, though statistical significance was not presented (Moffett and Zartman 1995). Also, at 15 Mg/ha application rate and above, the terminal cumulative infiltration flux and terminal infiltration on bare and vegetated plots were increased above measurements under untreated conditions (personal communication, David Wester, Texas Tech University, June 1997).

4. RISKS ASSOCIATED WITH CONTAMINANTS

4.1 GENERAL EXPOSURE ASSUMPTIONS AND JUSTIFICATION

Plants, soil invertebrates, and microorganisms that facilitate ecosystem-level processes, and wildlife and their foods are exposed to contaminants incorporated into soils. Toxicity is related to concentrations of chemicals in the exposure medium, which is soil. Thus, in this risk assessment potential effects of contaminants are determined on the basis of concentrations in soil rather than concentrations in sewage sludge or application rates. Little information exists regarding the long-term maximum incorporation of constituents of municipal sewage sludge into soils and the bioavailability of the contaminants. Moreover, the role of cumulative, multiple applications compared to single applications in determining the concentrations of contaminants in surface soils has not been established. In addition, studies comparing the relative retention of sludge-borne contaminants in surface soil after mixing or without mixing have not been undertaken.

This report presents an assessment of risks to wildlife, plants, soil invertebrates and microbial processes from sludge-borne contaminants at a single application rate, assuming a single application, and using a single soil/sludge mixing scenario. The assumptions are detailed below. Ecological risks are determined based on application of municipal sewage sludge containing concentrations of elements at the 95% upper confidence limit (UCL) about the mean from the 1988 National Sewage Sludge Survey. The current EPA cumulative loading limits for chemicals in municipal sewage sludge are expected to be associated with greater risks than those determined using the assumptions in this report.

Although Sections 4.1.4, Bioavailability; 4.1.5, Metal Speciation and Fate; and 4.1.6, Long-term considerations do not address the exposure assumptions directly, they do provide literature reviews that demonstrate in what circumstances and in what way the assumptions listed below are inaccurate. If an ecological endpoint in a specific ecosystem were of particular concern, methods could be developed to incorporate parameters controlling bioavailability. This assessment may be modified in the future according to such "bioavailability factors."

The exposure assumptions for the risk assessment are:

1) The concentration of contaminants in municipal sewage sludge is the 95% upper confidence limit of the mean contaminant concentration in the NSSS, presented in Table 4.1. It is assumed that the NSSS represents the current distribution of compositions of municipal sewage sludges.

2) A single application of sewage sludge at 40 Mg/ha is evaluated. Ecological risks may not be assumed, *a priori*, to be linearly associated with application rate.

3) Chemicals in biosolids are assumed to be completely and uniformly incorporated into the top 15 cm of soil. The time for mixing and contaminant release to occur is not considered in the assessment.

4) Plants, earthworms, arthropods, and wildlife are assumed to be exposed to the average concentration of sludge-borne contaminants in the top 15 cm rather than to potentially clean soil below this depth interval or to possibly high concentrations in the top few cm of soil.

5) De-watered sewage sludge is applied; thus, no water is assumed to be added to the soil.

6) There is no spatial heterogeneity in sludge application or in resulting soil concentration at an application site.

7) The bioavailability of soluble salts in soil is assumed to be higher than the bioavailability of the same concentrations of these chemicals in sewage sludge-treated soil.

8) The bulk density of soil is assumed to remain at the pre-sludge level.

Table 4.1. Summary Statistics for Concentrations of Contaminants in Sewage Sludge from the 1988 National Sewage Sludge Survey

Chemical of Concern	N	Detects	Minimum (mg/kg)	Geometric Mean (mg/kg)	Maximum (mg/kg)	Mean[a] (mg/kg)	95% UCL[a] (mg/kg)
Arsenic	177	147	0.20	6.07	258.7	10.41	13.05
Cadmium[b]	177	147	0.04	7.56	499	21.28	28.46
Copper	177	177	17.40	532.20	3060	704.9	774.92
Lead	177	159	3.90	120.62	1670	182	205.8
Mercury	177	139	0.10	2.50	47	3.83	4.5
Molybdenum	177	118	0.80	6.71	67.9	10.66	12.13
Nickel	177	148	2.00	35.15	976	69.8	84.6
Selenium	177	133	0.20	3.83	70	5.8	6.8
Zinc	177	177	37.80	965.15	68000	1886	2602
TCDD	177	78	6.00e-08	2.10e-06	0.00086	0.000032	0.00005
PCB-1248	177	19	0.02	0.02	5.2	0.12	0.17
PCB-1254	177	9	0.02	0.02	9.35	0.28	0.38
PCB-1260	177	10	0.02	0.02	4.01	0.09	0.13

[a] Mean and 95% upper confidence limit (UCL) on the mean calculated using product limit estimator method for incorporating non-detect values.
[b] Outlier observation of 8220 mg/kg deleted from data (EPA 1992a).

4.1.1 Concentration of Contaminants in Soil

Given the above assumptions, the equation for obtaining the concentration of a chemical in soil from that in municipal sewage sludge is:

$$C_{SS} = (C_S * 0.0267 / D_B) + C_{BS}$$

where

C_{SS} = concentration of contaminant in sludge-amended soil (mg/kg)
C_S = 95% upper confidence limit of concentration of chemical in sewage sludge from NSSS (mg/kg)
D_B = average bulk density of top 15 cm of soil in representative ecosystem (g/cm^3)
C_{BS} = ecosystem-specific background concentration of chemical in soil (mg/kg)

The quantity 0.0267 is the application rate expressed on a per volume basis (g/cm^3). It is calculated from the application rate (40 Mg/ha) divided by the soil depth (15 cm). All concentrations are expressed in dry weight. The concentrations of sludge-borne contaminants in soil in Douglas-fir forest, loblolly pine plantation, eastern deciduous forest, and semiarid rangeland are presented in Tables 4.2, 4.3, 4.4, and 4.5. The tremendous variability in chemical concentrations in sludge-treated soils, both from spatial variability in background concentrations of elements and variability in the composition of sewage sludge, is not expressed in the single example concentrations.

Table 4.2. Estimated concentrations of contaminants in soil (mg/kg dry wt.) in the Douglas-fir forest following sludge application. Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following one application of 40 Mg/ha

Chemical of Concern	Background Soil Concentration	Source	Addition of NSSS 95% UCL	mg/kg increase
Arsenic	2.6	Shacklette and Boerngen 1984[b]	2.917	0.317
Cadmium	1	Greenleaf-Jenkins and Zasoski 1986[a]	1.691	0.691
Copper	22	Greenleaf-Jenkins and Zasoski 1986	40.809	18.809
Lead	45	Greenleaf-Jenkins and Zasoski 1986	49.995	4.995
Mercury	0.051	Shacklette and Boerngen 1984	0.160	0.109
Molybdenum	3	Shacklette and Boerngen 1984	3.294	0.294
Nickel	14	Greenleaf-Jenkins and Zasoski 1986	16.053	2.053
Selenium	0.2	Shacklette and Boerngen 1984	0.365	0.165
Zinc	120	Greenleaf-Jenkins and Zasoski 1986	183.158	63.158
TCDD	0		0.000	0.000
PCB-1248	0		0.0041	0.0041
PCB-1254	0		0.0092	0.0092
PCB-1260	0		0.0032	0.0032

[a] From Pack Forest, WA

[b] Estimated from map, within 200 km of Pack Forest, WA; Pack forest measurements not available.

Table 4.3. Estimated concentrations of contaminants in soil (mg/kg dry wt.) in the Loblolly Pine Plantation following sludge application. Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following one application of 40 Mg/ha

Chemical of Concern	Background Soil Concentration	Source	Addition of NSSS 95% UCL	mg/kg increase
Arsenic	2	Shacklette and Boerngen 1984, W. Miller, pers. com.[a]	2.249	0.249
Cadmium	0.2	Shacklette and Boerngen 1984, W. Miller, pers. com.	0.743	0.543
Copper	10	Shacklette and Boerngen 1984, W. Miller, pers. com.	24.779	14.779

Lead	20	Shacklette and Boerngen 1984, W. Miller, pers. com.	23.925	3.925
Mercury	0.05	Shacklette and Boerngen 1984, W. Miller, pers. com.	0.136	0.086
Molybdenum	1	Shacklette and Boerngen 1984, W. Miller, pers. com.	1.231	0.231
Nickel	10	Shacklette and Boerngen 1984, W. Miller, pers. com.	11.613	1.613
Selenium	0.2	Shacklette and Boerngen 1984, W. Miller, pers. com.	0.330	0.130
Zinc	20	Shacklette and Boerngen 1984, W. Miller, pers. com.	69.624	49.624
TCDD	0		0.000	0.000
PCB-1248	0		0.0032	0.0032
PCB-1254	0		0.0072	0.0072
PCB-1260	0		0.0025	0.0025

[a] Prof. William Miller of the University of Georgia suggested these concentrations as the best estimates for the southern United States, based on Schacklette and Boerngen (1984).

Table 4.4. Estimated concentrations of contaminants in soil (mg/kg dry wt.) in the Eastern Deciduous Forest following sludge application. Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following one application of 40 Mg/ha

Chemical of Concern	Background Soil Concentration	Source	Addition of NSSS 95% UCL	mg/kg increase
Arsenic	4.1	Shacklette and Boerngen 1984[a]	4.448	0.348
Cadmium	0.2	Shacklette and Boerngen 1984	0.960	0.760
Copper	8.6	Ian Helm, pers. com.[b]	29.290	20.690
Lead	48.5	Ian Helm, pers. com.	53.995	5.495
Mercury	0.051	Shacklette and Boerngen 1984	0.171	0.120
Molybdenum	3	Shacklette and Boerngen 1984	3.324	0.324
Nickel	15	Shacklette and Boerngen 1984	17.259	2.259
Selenium	0.2	Shacklette and Boerngen 1984	0.382	0.182
Zinc	61.5	Ian Helm, pers. com.	130.973	69.473
TCDD	0		0.000	0.0000012
PCB-1248	0		0.0045	0.004539
PCB-1254	0		0.0101	0.010146
PCB-1260	0		0.0035	0.003471

[a] Estimate from map located in southwestern New Hampshire; Hubbard Brook Forest measurements not available.

[b] From Hubbard Brook Experimental Forest, NH.

Table 4.5. Estimated concentrations of contaminants in soil (mg/kg dry wt.) in the Semiarid Rangeland following sludge application. Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following one application of 40 Mg/ha

Chemical of Concern	Background Soil Concentration	Source	Addition of NSSS 95% UCL	mg/kg increase
Arsenic	2.2	S. Loftin, pers. com.	2.45	0.249
Cadmium	0.55	Harris-Pierce 1994[a]	1.093	0.543
Copper	4.84	Loftin 1996[b]	19.619	14.779
Lead	12.4	Loftin 1996	16.325	3.925
Mercury	0.054	Harris-Pierce 1994	0.140	0.086
Molybdenum	0.1	Harris-Pierce 1994	0.331	0.231
Nickel	11.5	Zartman 1993[c]	13.113	1.613
Selenium	1	S. Loftin, pers. com.	1.130	0.130
Zinc	39.84	Loftin 1996	89.464	49.624
TCDD	0		0.000	0.0000
PCB-1248	0		0.0032	0.0032
PCB-1254	0		0.0072	0.0072
PCB-1260	0		0.0025	0.0025

[a] From Meadow Springs Ranch, Larimer County, CO.
[b] From Sevilleta National Wildlife Refuge, NM; New Mexico values were used for background if available.
[c] From Sierra Blanca Ranch, TX.

The major assumptions of the equation are the 40 Mg/ha application rate and mixing in the top 15 cm of soil. The equation ignores potential changes in bulk density with sludge application and the mass contributed by the sludge. For this reason, if the background soil concentration of a contaminant is higher than the concentration in sludge, the equation still predicts that the concentration in soil increases with sludge amendment. The risk characterization corrects for this error when observed by concluding that sewage sludge should not increase ecological risks above those found at background soil concentrations.

Sewage sludge should have a lower bulk density than soil unless the soil has very high organic matter and porosity (Greenleaf-Jenkins and Zasoski 1986), but few measurements of the bulk density of sludge have been taken. An evaluation of bulk densities of soil following the application of sludge (Greenleaf-Jenkins and Zasoski 1986) found lowered bulk densities, but the application rate was much higher than in this study (512 Mg/ha). Wei et al. (1985) measured changes in bulk density with application of sludge and found a 3.5% decrease with application of 44.8 Mg/ha. Since bulk densities in this assessment are used to the tenths digit only (e.g., 1.1 g/cm^3), this percentage decrease is not considered.

4.1.2 Application Rate

A single application of biosolids at 40 Mg/ha is assumed in this assessment. Findings regarding ecological risks at this rate are assumed to be conservative conclusions for all single applications relying on an agronomic rate (i.e., no leaching of nitrate) and multiple applications

totaling less than 40 Mg/ha (e.g., 10 annual applications of 4 Mg/ha). Risks may increase at multiple application rates that total more than 40 Mg/ha. Data are not sufficient to distinguish exposures or effects at multiple applications from those at single applications.

The 40 Mg/ha application rate permits comparison to empirical studies on rangelands in New Mexico and Colorado where biosolids have been applied at 40 to 45 Mg/ha. The application rate is used for other ecosystems in this study for the purpose of comparability. In Douglas-fir forests, application rates that rely on an agronomic rate can be as high as 20 Mg/ha, with long-term application rates as low as 5 Mg/ha (Charles Henry, University of Washington, personal communication, April 8, 1996). At some sites, applications are proposed for every three years (e.g., Pack Forest, WA), but smaller communities are planning annual applications. Thus, long term multiple applications could easily exceed the 40 Mg/ha cumulative rate assumed in this study.

4.1.3 Soil Depth and Incorporation of Contaminants

For this risk assessment the conservative assumption was made that contaminants from biosolids are evenly distributed in the top 15 cm of soil. This was the assumption in the *Technical Support Document for Land Application of Sewage Sludge* (TSD; EPA 1992a) which stressed agricultural applications where sludge is mechanically incorporated. Nonagricultural applications of biosolids are expected to result in slower incorporation and leaching of elements than agricultural applications, given that the former type of application is usually not mechanically mixed. In forests and particularly in rangeland (because of low precipitation), the mixing of sludge-amended contaminants with soil would be expected to be much slower than in agriculture. For example, in Colorado rangeland the concentrations of some elements (e.g., As, Se) in surface soil did not increase following sludge application (Harris-Pierce 1994). Without mass balance estimates of leaching, runoff, and plant-uptake following sludge application, it is impossible to quantify the losses of sludge contaminants from the top 15 cm of soil. (One study of the loss of metals fifteen years after sludge application in an agricultural system, found approximately 40% of sludge-amended zinc and copper, 30% of cadmium, and half of the mercury lost from the topsoil (McBride et al. 1997), but such mass balance studies are rare.) Thus, in the forests and rangeland ecosystems of this study, it is assumed that no losses have occurred from the top 15 cm of soil.

In addition to the precedent set by the TSD (EPA 1992a), other logic and data support the use of the 15 cm depth. In Fulton County, IL, 80 to 100% of the mercury applied to the soil for two decades is present in the top 15 cm of soil (Granato et al. 1995). At one location at the University of Washington's Pack Forest, metals from sewage sludge disked into soil and seeded with rye grass largely remained in the top 15 cm of soil after 16 years (Brallier et al. 1996). McGrath (1987) reviewed long-term (3-13 years) field applications of sludge in agriculture and found that little leaching of metals below the plough layer (27 cm) occurred; four studies cited recoveries of 65% to over 90% in the topsoil. Gunderson et al. (1995) found that constituents of sludge applied to the Oak Ridge Reservation in Tennessee are largely retained in the top 15 cm of soil. In addition to these rationales, the 15 cm depth corresponds to the minimum (or sometimes the only) depth of sampling in field applications of sludge that are used in the analyses in the following chapters. Soil-to-biota uptake models were usually based on studies that measured soil concentrations in the top 10 to 20 cm of soil.

Some studies suggest that concentrations of contaminants from sludge are found at higher concentrations in the top 5 or 10 cm of soil than at lower depths (Sidle and Sopper 1976, Harris-Pierce 1994). Indeed, although Harris-Pierce (1994) does not provide mass-balances for contaminants added to soil in sludge, up to 90% of the added metals reside in the top 8 cm of soil in Colorado grasslands 2 years after application. Sidle and Sopper (1976) also show that the concentration of cadmium is higher in the top 5 cm of soil than in the soil below, when irrigated with wastewater. In a study on Colorado rangeland, zinc increased in residual surface biosolids for two years following application, perhaps because of the decrease in easily decomposable organic matter (Pierce 1994). Few other studies have analyzed residual biosolids for contaminants.

If all of the sludge were mixed into the top 5 cm, organisms remaining in that portion of surface soil would have higher exposure to contaminants than if even distribution in the top 15 cm is assumed. Arthropods living near the soil surface, litter-feeding earthworms and their predators would be affected. On the other hand, exposures to plants and many earthworms are not confined to the top several cm of soil; thus their predicted exposures would probably be similar under a 5 or 15 cm mixing scenario. We have assumed the 15 cm depth of mixing for the reasons mentioned above, and since few studies show all of the sludge incorporated in the top several cm of soil.

4.1.4 Bioavailability

It is generally assumed that heavy metals applied to soils as soluble salts are more bioavailable than those originating from sewage sludge (Bell et al. 1991). Thus, studies of phytotoxicity to plants or uptake by plants where salts are added to soils represent conservative estimates of results if sewage sludge was the source of the contaminants. Bioavailability is addressed implicitly in the derivation of soil-to-plant uptake factors (Section 4.2.1.2), since field studies with biosolids amendments were utilized. It may be erroneous to assume that chemicals originating from sewage sludge are less available to earthworms, arthropods, or wildlife ingesting soil than other forms of contaminants. Studies comparing the bioavailability of metals from different sources to these organisms have not been undertaken.

Numerous studies have been undertaken to determine the forms of metals in sewage sludge (soluble, precipitated, co-precipitated in metal oxides, associated with biological residues) (Lake et al. 1984). In soil, metal cations are most available to plants as simple or complexed ions in soil solution, easily exchangeable ions, or as organically bound chemicals (Soon and Bates 1982). Because of the variability of sewage sludge composition, and the general lack of data concerning the relationship of various fractions of sludge to toxicity and/or uptake, the soil chemistry of sewage sludge is not explicitly considered in the estimation of ecotoxicity or uptake by biota.

The bioavailability of metals to plants from soils (Sadiq 1985) or sewage sludge at various application rates (Fresquez et al. 1990a) is often correlated with extractability with diethylene triamine pentaacetic acid (DTPA). However, most available studies concerning uptake by and toxicity to plants and invertebrates in soil use total extractions with strong acids. Of the target ecosystems of this ecological risk assessment, measurements of plant toxicity of DTPA-extractable contaminants exist only for the rangeland; thus this line of evidence is used for toxicity in the rangeland ecosystem only. However, criticisms of the use of DTPA to estimate bioavailability are common. DTPA is not a good predictor of uptake of metals into

plants from sewage sludge with low metal content (Barbarick and Workman 1987). In addition, the DTPA estimate of biovailability has been observed not to work well for arsenic (Sadiq 1986) or nickel or lead (Sadiq 1985) in corn or for comparison of bioavailability to plants between soils of differing pH (Miles and Parker 1979).

The addition of municipal sewage sludge to soils is generally thought to decrease the bioavailability of metals in the soils. Some exceptions are observed, such as the decrease in sorption of lead following the application of sludge to a soil in India (Hooda and Alloway 1994). The major means by which sewage sludge may increase the bioavailability of metals is through a potential decrease in soil pH with biosolids amendment. Hooda and Alloway (1994) observed an increase in soluble cadmium, chromium, cobalt, copper, lead, nickel, vanadium, and zinc as pH decreased below 5. Such a decrease in pH may be due to: a low sludge pH, leaching of organic acids, sludge decomposition and production of carbonic acid, or nitrification of ammonium (Aguilar et al. 1994). Soil carbonates may neutralize these effects with time (Aguilar et al. 1994). A decrease in soil pH attributable to sewage sludge has not always altered the bioavailability of heavy metals in sludge-treated soils; for example, a decrease in pH from 7.8 to 7.4 in the Rio Puerco valley grassland did not significantly increase the solubility of cadmium or lead (Fresquez et al. 1990).

Other factors that have increased soluble metal contents in soils include increasing concentrations of chloride ions and increases in pH above 7 for vanadium, chromium, copper, and lead (Hooda and Alloway 1994).

4.1.5 Mercury Speciation and Fate

Mercury is the principal assessment chemical for which different species have been observed to be associated with varying levels of toxicity. Forms include mercuric chloride, mercuric sulfide, elemental mercury, and methyl or dimethyl mercury. A study at a mercury-contaminated sewage treatment plant observed total sludge mercury concentrations ranging from 17 to 21 mg/kg and methyl mercury concentrations ranging from 0.001 to 0.008 mg/kg (Gilmour and Bloom 1994). Thus, in this study methyl mercury comprised less than 0.1 percent of the total mercury in sewage sludge. Carpi et al. (1997) found a similar fraction of methyl mercury in sludge from Oak Ridge, TN (0.03 to 0.11%). The NSSS did not survey treatment plants for various mercury species, so the distributions of various mercury concentrations in biosolids is unknown. However, the assessment of risks to wildlife assumes that the majority of mercury from sludge is taken up in inorganic form.

Results from Oak Ridge National Laboratory and Cornell University suggest that sludge-amended soils can contribute to significant emissions of methyl mercury and inorganic mercury from sludge-amended soils (Carpi and Lindberg 1997, Carpi et al. 1997). The source of the methyl mercury is probably the quantity of the chemical in the sewage sludge amendment. Elemental mercury emission is caused by a photo-stimulated reduction of oxidized mercury. These transformations of mercury have not explicitly been considered in studies of the uptake of mercury by biota, but the processes are implicitly included in empirical toxicity and uptake studies involving sludge amendments.

4.1.6 Long Term Considerations

The bioavailability of constituents in sewage sludge is affected by the potential degradation or transformation of the constituents and the fate of the sludge matrix relative to the sorption of chemicals. The rate of leaching and runoff compared to the rate of infiltration is particularly important for predicting the concentrations of constituents of sludge with time.

Organic chemicals may be degraded in soil. Berthouex (1991) observed the degradation of 2-, 3-, and 4-chlorinated PCB isomers from farmland according to first-order kinetics that are independent of sludge loading rate, PCB concentration or application frequency. However, the more chlorinated isomers remained after five years. The degradation of organic constituents in sludge are affected by whether the sludge is anaerobically-digested or aerobically-digested (Petronio et al. 1993).

The sorptive capacity of soils for metals may be altered not only by the addition of sewage sludge but by the decomposition of the sludge. Much evidence supports the view that the binding of metals by sewage sludge is primarily due to inorganic constituents of sewage sludge (EPA 1995). The likely sorptive residues include phosphates, silicates, and iron, aluminum, and manganese oxides (McBride 1995). However, some researchers suggest that the availability of metals in soil may increase as the organic constituents of sludge decompose (McBride 1995); this view has been termed the "time-bomb theory" (EPA 1995). Indeed, much evidence suggests that cadmium is much less bioavailable when bonded to organic constituents than inorganic constituents of sewage sludge (McBride 1995). McBride (1995) also notes that the carbonates or sulfides in sludge are unlikely to reduce bioavailability over the long term.

Definitive studies examining the long term (several decades or longer) bioavailability of sludge-borne contaminants have not been conducted, and this assessment reflects that uncertainty. To determine risks to all ecological endpoints in this assessment, lines of evidence with different bioavailability assumptions are weighed. Some lines of evidence assume that the entire contaminant mass in sludge-amended soil may someday become available. Others assume that bioavailability or maximum exposure may be estimated by the uptake of contaminants by biota during the sampling intervals (months to a few years) of existing studies.

4.2 RISKS TO WILDLIFE

4.2.1 Exposure Assessment for Wildlife

Terrestrial wildlife endpoints may be exposed to and potentially affected by contaminants in sludge through the food web or through incidental ingestion of soil while foraging or grooming. A conceptual model outlining these relationships is presented in Figure 1.4 and described in Sect. 1.5.3.

In this assessment, exposure to contaminants through ingestion of food and soil was estimated for four trophic groups within each ecosystem: mammalian ground invertebrate feeders, avian invertebrate feeders, carnivores, and herbivores. Specific wildlife endpoint species, by ecosystem, are listed in Table 1.4. Exposure estimates were calculated for each endpoint using estimated contaminant concentrations in the top 15 cm of soil (Sect. 4.1) and estimated contaminant concentrations in food items. Contaminant concentrations in wildlife foods were estimated using soil-biota uptake models (Sect. 4.2.1.2).

Two types of exposure estimates were generated. The first, or screening exposure estimate, is based upon conservative point-estimates of parameters in the exposure model. This estimate is compared to No-Observed Adverse Effect Levels (NOAELs) and Lowest-Observed Adverse Effect Levels (LOAELs) for each of the four trophic groups (Sect. 4.2.2) and is used to identify Contaminants of Potential Ecological Concern (COPECs) for each ecosystem. Monte Carlo exposure estimates are then calculated for the COPECs. Monte Carlo estimates incorporate the variability of parameters in the exposure model and result in exposure distributions that may be used to estimate the likelihood of experiencing exposures in excess of NOAELs or LOAELs.

4.2.1.1 Oral ingestion exposure model

Oral exposure to contaminants experienced by wildlife may come from multiple sources. Terrestrial wildlife may consume contaminated food (either plant or animal), drink contaminated water, or ingest soil. Soil ingestion may be incidental while foraging or grooming or purposeful to meet nutrient needs. Because exposure through water is outside the scope of this assessment, the total oral exposure experienced by an individual may be described as:

$$E_{total} = E_{food} + E_{soil}$$

Where:

E_{total}	=	total exposure from all pathways
E_{food}	=	exposure from food consumption
E_{soil}	=	exposure from soil consumption

For exposure estimates to be useful in the assessment of risk to wildlife, they must be expressed in terms of a body weight-normalized daily dose or mg contaminant per kg body weight per day (mg/kg/d). Exposure estimates expressed in this manner may then be compared to toxicological benchmarks for wildlife, such as those derived by Sample et al. (1996b), or to doses reported in the toxicological literature. Estimation of the daily contaminant dose an individual may receive from a particular medium for a particular contaminant may be calculated using the following equation:

$$E_j = \sum_{i=1}^{m} \sum_{k=1}^{n} p_{ik} \left(\frac{IR_i \times C_{ijk}}{BW} \right)$$

where:

E_j	=	total exposure to contaminant (j) (mg/kg/d)
m	=	total number of ingested media (e.g., plant, animal, soil, etc.)
n	=	number of types of medium (i) consumed (unitless)
IR_i	=	ingestion rate for medium (i) (kg/d)
p_{ik}	=	proportion of type (k) of medium (i) consumed (unitless)
C_{ijk}	=	concentration of contaminant (j) in type (k) of medium (i) (mg/kg)
BW	=	body weight of endpoint species (kg)

4.2.1.2 Soil-biota bioaccumulation models

To estimate the concentrations of contaminants in foods consumed by wildlife from concentrations in sludge-amended soil, soil-biota bioaccumulation models for plants,

earthworms, arthropods and small mammals are needed. Soil-biota bioaccumulation models may consist of simple uptake factors (ratio of the concentration of a given contaminant in biota to that in the soil where the organism resides) or other empirical models, such as log-linear regression models. In general, estimates generated by uptake factors are more conservative (overestimate bioaccumulation, particularly at high chemical concentrations in soil) than those generated by log-linear models (Sample et al. 1998a, 1998b). In this risk assessment, conservative point-estimates of the uptake factors (90th percentile) for different chemicals in soil are used in the screening assessment, and distributions of results from the log-linear regressions are inputs for Monte Carlo analyses in the definitive assessment (see risk characterization, Sect. 4.2.3.1). Because soil and contaminant data used to develop uptake models are generally expressed as dry weight, biota contaminant concentrations estimated from uptake factors are also in dry weight.

Plants

Plant and co-located soil concentrations from sewage sludge-amended sites, including measurements in Colorado rangeland, Douglas-fir forest, and an old field in Ohio were compiled. Measurements where soil concentrations were not measured as total concentrations (e.g. New Mexico rangeland, Fresquez et al. 1990) were excluded. These data from sludge-treated sites were determined to be insufficient to use in the derivation of soil-plant uptake models (Appendix 1). Chemical concentrations in plants and soils were typically from fewer than five studies and often from only one ecosystem. For example, all plant uptake data for arsenic was from one study in the semi-arid rangeland (Harris-Pierce 1994). In addition, the range of measured, sludge-treated soil concentrations in a single field study was typically narrow.

Empirical models (uptake factors and log-linear regressions) were previously developed for all contaminants in this study except molybdenum, PCBs, and TCDD (BJC 1998). These models were derived from published chemical data from field studies; very few data were from sewage sludge-amended soils. Sources of contamination included mine waste, smelter deposits, vehicle and other urban emissions, wastewater effluent, compost, fertilizer, dredged material, fly ash, flue dust, and sewage sludge. Additional concentrations taken at background locations were used. Data from the field studies utilized in the derivation of models in BJC (1998) may be found in an Appendix of that document. Summary statistics for these uptake factors and log-linear regressions are presented in Table 4.6. To determine if bioaccumulation differed for plants grown in sewage sludge-amended soils, contaminant concentration in soil was plotted against contaminant concentration in plants for each contaminant for both sewage sludge field studies from Appendix 1 and studies from BJC (1998) (Figs 4.1, 4.2). Almost all observations from sludge studies fell within the 95% prediction interval of the loglinear regression models fit to plant bioaccumulation data from primarily non-sludge sources (Figs 4.1, 4.2). Observations from studies of the uptake of contaminants from sludge-amended soils that were outside the 95% prediction interval fell below the lower limit. These results indicate that existing bioaccumulation models derived from field contaminant data can be used to generate reasonable or conservative estimates of contaminant bioaccumulation by plants grown in sewage sludge-treated soils.

Because bioaccumulation models for molybdenum by plants were not available, models were developed based on data from field studies of sewage sludge application. Data were available only for the arid range ecosystem. Data used for model development are reported in Appendix 1. A scatterplot of the molybdenum concentration in soil versus the concentration in plants is displayed in Fig. 4.3. The loglinear regression model was significant ($p<0.001$) with

Table 4.6. Soil-to-Plant Uptake Models

	Summary Statistic for Uptake Factors				Parameters for loglinear uptake models[a]		
Analyte	**Minimum**	**Median**	**90th Percentile**	**Maximum**	**Slope±SE**	**Intercept±SE**	**Source**
Arsenic	0.00006	0.03752	1.1034	9.0741	0.564±0.125	-1.991±0.431	BJC 1998
Cadmium	0.0087	0.58571	3.25	22.8788	0.546±0.042	-0.475±0.088	BJC 1998
Copper	0.0011	0.12432	0.625	7.4	0.394±0.043	0.668±0.212	BJC 1998
Lead	0.00011	0.0388	0.4675	10.6011	0.561±0.072	-1.328±0.350	BJC 1998
Mercury	0.00145	0.65217	5	12.2301	0.544±0.037	-0.996±0.121	BJC 1998
Molybdenum	0.132	0.528	7.06	37.5	0.237±0.046	0.327±0.077	Appendix 1
Nickel	0.00217	0.01786	1.4113	22.2143	0.748±0.093	-2.223±0.471	BJC 1998
Selenium	0.02	0.67189	3.0119	77	1.104±0.067	-0.677±0.140	BJC 1998
Zinc	0.00855	0.36616	1.8203	34.2857	0.554±0.045	1.575±0.278	BJC 1998
PCBs(undefined)	0.000	0.000	0.000	0.000	NA	NA	O'Connor et al. 1990
TCDD	0.000	0.000	0.000	0.000	NA	NA	Muller et al. 1993

[a] model: ln(plant)=intercept+slope (ln[soil]).

r^2=.29. Summary statistics for the plant bioaccumulation models for molybdenum are presented in Table 4.6.

Based on the results of research by O'Connor et al. (1990) and Muller et al. (1993), the bioaccumulation of PCBs and TCDD by plants was assumed to be negligible. Consequently uptake factors for these contaminants were set to 0 (Table 4.6)

Arthropods

Because bioaccumulation models for contaminants by arthropods were not available in the literature, they were developed using methods similar to those derived for earthworms by Sample et al. (1998a). These models were derived from published data from field studies and contain no data concerning bioaccumulation by plants grown in sewage sludge amended soils. Data used for model development are reported in Appendix 2. Summary statistics for these models are presented in Table 4.7. Loglinear regression models were significant ($p<0.05$) for all contaminants except zinc ($p=0.08$). r^2 values ranged from 0.16 for copper to 0.88 for PCBs. To determine if bioaccumulation differed for arthropods associated with sewage sludge-amended soils, contaminant concentration in soil was plotted against contaminant concentration in arthropods for each contaminant for both sewage sludge studies and non-sludge studies (Fig. 4.4). No data for PCBs in arthropods from sewage sludge-treated sites were found. All observations from municipal sewage sludge studies fell within the 95% prediction interval of the loglinear regression models fit to non-sludge plant bioaccumulation data (Fig. 4.4). These results indicate that bioaccumulation models based on samples taken at metal-contaminated sites can be used to generate suitable estimates of contaminant bioaccumulation by arthropods associated with sewage sludge-amended soils.

Earthworms

Bioaccumulation models for earthworms for all contaminants except molybdenum were obtained from Sample et al. (1998a). No data concerning bioaccumulation of molybdenum by earthworms were found. These models were derived from published data derived from field studies and are based primarily on data from non-sludge sites. Summary statistics for these models are presented in Table 4.8.

Small Mammals

Bioaccumulation models for small mammals for all contaminants except molybdenum and PCBs were obtained from Sample et al. (1998b). No suitable data concerning bioaccumulation of molybdenum or PCBs by small mammals was found. These models were derived from published concentration data from field studies at non-sewage sludge amended sites. With the exception of TCDD, no data concerning bioaccumulation of contaminants by small mammals at sludge-amended sites were found. The TCDD models were based primarily on data from a field application of TCDD contaminated paper mill sludge (ERT 1987). Summary statistics for these models are presented in Table 4.9.

Table 4.7 Soil-to-Arthropod Uptake Models

	Summary Statistic for Uptake Factors				Parameters for loglinear uptake models[a]		
Chemical	Minimum	Median	90th Percentile	Maximum	Slope±SE	Intercept±SE	Source
Cadmium	0.0205	0.556	3.064	24	0.430±0.066	0.666±0.195	Appendix 2
Copper	0.0049	0.667	3.185	14.3846	0.296±0.076	2.419±0.366	Appendix 2
Lead	0.00031	0.046	0.455	1	0.594±0.057	-0.558±0.395	Appendix 2
Zinc	0.0103	0.357	2.902	9.7844	0.102±0.057	5.245±0.417	Appendix 2
PCBs (1254)	0.072	0.149	0.191	0.191	0.866±0.185	-1.187±1.156	Paine et al. 1993

[a] model: ln(plant)=intercept+slope (ln[soil]).

Table 4.8. Soil-to-Earthworm Uptake Models

	Summary Statistic for Uptake Factors				Parameters for loglinear uptake models[a]		
Chemical	Minimum	Median	90th Percentile	Maximum	Slope±SE	Intercept±SE	Source
Arsenic	0.006	0.224	0.523	0.925	0.706±0.169	-1.421±0.327	Sample et al. 1998a
Cadmium	0.253	7.708	40.690	190.000	0.795±0.037	2.114±0.079	Sample et al. 1998a
Copper	0.002	0.515	1.531	5.492	0.264±0.040	1.675±0.141	Sample et al. 1998a
Lead	0.000	0.266	1.522	228.261	0.807±0.044	-0.218±0.245	Sample et al. 1998a,
Mercury	0.030	1.693	20.625	33.000	0.118±0.089	-0.684±0.198	Sample et al. 1998a
Molybdenum							no data available
Nickel	0.033	1.059	4.730	7.802	-0.260±0.196	3.677±0.635	Sample et al. 1998a
Selenium	0.300	0.985	1.340	13.733	0.733±0.256	-0.075±0.194	Sample et al. 1998a
Zinc	0.025	3.201	12.885	49.510	0.328±0.024	4.449±0.132	Sample et al. 1998a
PCBs(undefined)	0.000	6.667	15.909	65.227	1.361±0.088	1.410±0.210	Sample et al. 1998a
TCDD	1.191	11.011	22.229	42.068	1.182±0.074	3.533±0.810	Sample et al. 1998a

[a] model: ln(earthworm)=intercept+slope (ln[soil]).

Table 4.9. Soil-to-Small Mammal Uptake Models[a]							
	Summary Statistic for Uptake Factors				Parameters for loglinear uptake models[b]		
Chemical	**Minimum**	**Median**	**90th Percentile**	**Maximum**	**Slope±SE**	**Intercept±SE**	**Source**
Arsenic	0.000	0.003	0.015	0.071	0.8188±0.1043	-4.8471±0.4347	Sample et al. 1998b
Cadmium	0.015	0.333	3.991	69.561	0.4865±0.1016	-0.4306±0.1809	Sample et al. 1998b
Copper	0.004	0.196	1.045	1.398	0.1444±0.0285	2.0420±0.1301	Sample et al. 1998b
Lead	0.003	0.105	0.286	2.659	0.4422±0.0497	0.0761±0.2524	Sample et al. 1998b
Mercury	0.018	0.054	0.192	1.046	-2.2764±2.6962	-4.8666±1.7959	Sample et al. 1998b
Molybdenum							no data available
Nickel	0.000	0.248	0.589	1.143	0.4658±0.0729	-0.2462±0.1970	Sample et al. 1998b
Selenium	0.000	0.162	1.187	1.754	0.3764±0.1125	-0.4158±0.2090	Sample et al. 1998b
Zinc	0.005	0.772	2.688	16.364	0.0738±0.0194	4.4713±0.1122	Sample et al. 1998b
PCBs(undefined)							no data available
TCDD	0.3070	1.0667	2.2000	2.2000	1.0993±0.1852	0.8113±1.8493	Sample et al. 1998b

[a] Values represent general uptake models and not trophic-group-specific models.
[b] model: ln(small mammal)=intercept+slope (ln[soil]).

The bioaccumulation models are intended to represent a wide variety of sites, soils, and organisms. A high degree of uncertainty exists if median uptake factors or regressions are applied without retaining information about uncertainty bounds. The presence of sewage sludge may alter the bioavailability of metals and organic contaminants to biota (Sect. 4.1.4). For example, with the use of an uptake model, metals present at background levels in soils are treated the same as those added in sewage sludge. Models could be modified to incorporate soil characteristics.

4.2.1.3. Life history parameters and exposure assumptions for endpoint species

To estimate exposures, life history parameters and exposure assumptions must be defined for each endpoint species. These parameters and assumptions include body weight, food and water consumption rates, and diet composition. These parameters and assumptions for each endpoint species in each ecosystem are outlined in Table 4.10.

The soil-biota contaminant uptake factors presented in Section 4.2.1.2 were derived from dry weight concentrations in soil and biota; therefore the estimates of concentrations in wildlife foods generated by their use are also in terms of dry weight concentrations. Because the food ingestion rates for all endpoint species (except mule and black-tailed deer) are in wet weight, for comparability, all estimates of contaminant concentrations in food items must be expressed as wet weight concentrations. Dry weight concentrations were converted to wet weight concentrations using the following equation:

$$C_{ww} = C_{dw} \times p_{dry\ matter}$$

where:

C_{ww} = estimated wet weight contaminant concentration (mg/kg) in food
C_{dw} = dry weight contaminant concentration (mg/kg) estimated with uptake factors
$p_{dry\ matter}$ = food type specific proportion dry matter content

Values for $p_{dry\ matter}$ for food types consumed by endpoint species considered in this assessment were obtained from EPA (1993) and are presented in Table 4.11.

Table 4.10. Life History Parameters and Exposure Assumptions for Wildlife Endpoint Species

Species	Scientific Name	Parameter	Value	Citation/comment
Trowbridge Shrew	*Sorex trowdridgii*	Body Weight	median = 0.00449 kg minimum=0.004 kg maximum=0.0055 kg	Silva and Downing 1995
		Food Ingestion Rate	median = 0.0052 kg/d minimum = 0.0041 kg/d maximum = 0.0064 kg/d	George 1989 (assumed to be wet wt.)
		Soil Ingestion Rate	13% of diet or 0.00035 kg/d (dry wt.)	assumed proportional to that for short-tailed shrew
		Diet Composition	100% arthropod	George 1989
Short-tailed Shrew	*Blarina brevicauda*	Body Weight	0.015±0.00078 kg	Schlessinger and Potter 1974 (mean±std)
		Food Ingestion Rate	median=0.009 kg/d minimum=0.008 kg/d maximum=0.01 kg/d	Buckner 1964, Barrett and Stueck 1976 (wet wt.)
		Soil Ingestion Rate	13% of diet or 0.00117 kg/d (dry wt.)	Talmage and Walton 1993
		Diet Composition	31% earthworms reported; 100% earthworms assumed for assessment	Whitaker and Ferraro 1963
Desert Shrew	*Notiosorex crawfordi*	Body Weight	median=0.0052 kg minimum=0.0044 kg maximum=0.0063 kg	Silva and Downing 1995
		Food Ingestion Rate	0.0031 kg/d (wet wt.)	Estimated using median body weight and allometric equation from Nagy (1987) and water content in food items from EPA (1993a)[a]
		Soil Ingestion Rate	13% of diet or 0.0004 kg/d (dry wt.)	assumed proportional to that for short-tailed shrew
		Diet Composition	100% arthropod	Armstrong and Jones 1972
Red Fox	*Vulpes vulpes*	Body Weight	4.5 kg	Storm et al. 1976

Species	Scientific Name	Parameter	Value	Citation/comment
		Food Ingestion Rate	0.45 kg/d (wet wt.)	Vogtsberger and Barret 1973; Sargent 1978
		Soil Ingestion Rate	0.0126 kg/d (dry wt.) (soil consumption 2.8% of diet)	Beyer et al. 1994
		Diet Composition	81%small mammals, 10% fruits, 9% invertebrates	Hockman and Chapman 1983
Kit Fox	*Vulpes macrotis*	Body Weight	2.1 kg	McGrew 1979
		Food Ingestion Rate	0.325 kg/d (wet wt.)	Estimated using allometric equation from Nagy (1987) and water content in food items from EPA (1993a)[b]
		Soil Ingestion Rate	0.006 kg/d (dry wt.)	assumed proportional to that for red fox
		Diet Composition	100% small mammals	McGrew 1979
Mule Deer and Black-tailed Deer	*Odocoileus hemionus*	Body Weight	median=66.2 kg minimum=59.4 kg maximum=72.9 kg	Alldredge et al. 1974
		Food Ingestion Rate	median=1.45 kg/d (dry wt.) minimum=1.106 kg/d (dry wt.) maximum=2.87 kg/d (dry wt.)	Alldredge et al. 1974
		Soil Ingestion Rate	median=0.0161 kg/d (dry wt) minimum=0.0079 kg/d (dry wt) maximum=0.0296 kg/d (dry wt)	Arthur and Alldredge 1979
		Diet Composition	100% plant foliage	Anderson and Wallmo 1984
		Comments	species pooled as black-tailed deer is frequently viewed as subspecies of mule deer	Anderson and Wallmo 1984
White-tailed Deer	*Odocoileus virginianus*	Body Weight	56.5 kg	Smith 1991
		Food Ingestion Rate	1.74 kg/d (wet wt.)	Mautz et al. 1976

Species	Scientific Name	Parameter	Value	Citation/comment
		Soil Ingestion Rate	0.0348 kg/d (dry wt.) (soil consumption <2% of diet; 2% assumed)	Beyer et al. 1994
		Diet Composition	100% plant	Smith 1991
American Robin	*Turdus migratorius*	Body Weight	median=0.077 kg minumum=0.0635 kg maximum=0.103 kg	Dunning 1993
		Food Ingestion Rate	median=0.093 kg/d (wet wt) minimum=0.0685 kg/d (wet wt) maximum=0.117 kg/d (wet wt)	Hazelton et al. 1984; Skorupa and Hothem 1985.
		Soil Ingestion Rate	2.1% of diet or 0.0019 kg/d (dry wt.)	Sample and Suter 1994
		Diet Composition	37.5% invertebrate (mostly insect) and 62.5% plant (mostly fruits)	Wheelwright 1986
Western Meadowlark	*Sturnella neglecta*	Body Weight	0.1007 kg	Dunning 1993
		Food Ingestion Rate	0.0028 kg/d (wet wt.)	Sample et al. 1997
		Soil Ingestion Rate	0.0026 kg/d (dry wt.)	assumed proportional to that for American Robin
		Diet Composition	70% insects and 30% seeds	Bent 1958

[a]Estimation of food ingestion rate for desert shrew using allometric equation of Nagy (1987):

food ingestion (kg/d dry wt) = $0.067(BW)^{0.822} = 0.067(0.0052\ kg)^{0.822} = 0.0009$ kg/d dry wt.

assuming a diet of 100% invertebartes and with mean water content of 71.3% (mean for earthworms, crickets, and beetles; EPA 1993a)

food ingestion (kg/d wet wt.) = 0.0009 kg/d / proportion dry matter in food = 0.0009 kg/d / (1-0.713) = 0.0031 kg/d wet wt.

[b]Estimation of food ingestion rate for kit fox using allometric equation of Nagy (1987):

food ingestion (kg/d dry wt) = $0.067(BW)^{0.822} = 0.067(2.1\ kg)^{0.822} = 0.104$ kg/d dry wt.

assuming a diet of 100% small mamals and with mean water content of 68% (EPA 1993a)

food ingestion (kg/d wet wt.) = 0.104 kg/d / proportion dry matter in food = 0.104 kg/d / (1-0.68) = 0.325 kg/d wet wt.

Table 4.11. Water and dry matter content of food types consumed by endpoint species. All values from EPA (1993a)

Food Taxa	Percent Water (mean±STD)	Proportion Water	Proportion Dry Matter	Comments
Arthropods	65±7.7%	0.65	0.35	Mean of values for crickets and beetles
Earthworms	84±1.7%	0.84	0.16	
Small Mammals	68±1.6%	0.68	0.32	
Plant-Fruits	77±3.6%	0.77	0.23	
Plant-Seeds	9.3±3.1%	0.093	0.907	
Plant-Foliage	50%	0.5	0.5	Reported values ranged from 77-88% for dicot foliage and 7-10% for dry grass. 50% assumed to be conservative.

4.2.1.4. Screening exposure estimates

Conservative estimates of exposure to inorganic and organic contaminants were calculated to determine Contaminants of Potential Ecological Concern (COPECs) for all four endpoint species in each of the four ecosystems. For each endpoint species, two exposure scenarios were considered:

1) exposure to contaminants in soil at concentrations resulting from a one time application of 40 Mg/ha containing the 95% UCL about the mean concentration from the 1988 NSSS; and

2) exposure to background soil concentrations in the absence of any sewage sludge addition.

Exposure estimates from scenario 1 serve to delineate hazards that may be expected from a high-metal-containing sewage sludge generated in the United States and applied at 40 Mg/ha. Exposure estimates from scenario 2 provide an indication of the contribution of background concentrations to the risk estimates obtained by scenario 1. Estimated soil concentrations resulting from these scenarios for the Douglas-fir forest, loblolly pine plantation, eastern deciduous forest, and semiarid rangeland are presented in Tables 4.2, 4.3, 4.4, and 4.5, respectively. Exposure estimates were calculated using the assumptions in Section 4.1.

Exposure estimates were calculated for each endpoint species using the exposure equation (Section 4.2.1.1.) and the life history parameters from Table 4.10. To be conservative for the purpose of screening exposure estimates, contaminant concentrations in wildlife foods were estimated using the 90th percentile of the soil-biota uptake factors presented in Section 4.2.1.2. With the exception of food for mule and blacktailed deer, estimated dry weight contaminant concentrations in food were converted to wet weight estimates using dry matter content of food types from Table 4.11. Food concentrations for mule and blacktailed deer were not converted to wet weight because the food consumption rate value for these endpoints is expressed as dry weight (Table 4.10).

The diet of short-tailed shrews was assumed to consist of 100% earthworms (Table 4.10). Consequently, the earthworm uptake factors and earthworm dry matter content were used for all contaminants. The invertebrate portions of the diets of Trowbridge shrews, desert shrews, western meadowlarks, American robins, and red fox were assumed to be comprised exclusively of arthropods. Therefore, arthropod uptake factors were used for all contaminants for which they were available (Cd, Cu, Pb, Zn, and PCBs); earthworm uptake factors were used for the remaining contaminants (As, Hg, Ni, Se, and TCDD). Regardless of whether the uptake factor used was for earthworms or arthropods, the dry matter content for arthropods was used to estimate the fresh weight of the invertebrate portion of the diet of these species.

Screening exposure estimates for wildlife endpoints within the Douglas-fir forest are presented in Tables 4.12 through 4.15, endpoints within the loblolly pine plantation ecosystem are presented in Tables 4.16 through 4.19, endpoints within the eastern deciduous forest are presented in Tables 4.20 through 4.23, and endpoints within the semiarid rangeland ecosystem are presented in Tables 4.24 through 4.27.

4.2.1.5. Monte Carlo exposure estimates

Point estimates for the input parameters in the exposure model do not take into account the variation and uncertainty associated with the parameters and therefore may overestimate the contaminant exposure that wildlife endpoint species receive. The influence of variation in exposure parameters on the final estimate may be evaluated through the use of Monte Carlo simulation.

Monte Carlo simulation is a resampling technique frequently used in uncertainty analysis in risk assessment (Hammonds et al. 1994). In practice, distributions are assigned to input parameters in a model, and the model is recalculated many times to produce a distribution of output parameters (e.g., estimates of contaminant exposure). Each time the model is recalculated, a value is selected from within the distribution assigned for each input parameter. As a result, a distribution of exposure estimates is produced that reflects the variability of the input parameters.

Simulations were performed for each contaminant and endpoint within an ecosystem where comparison of point estimates of exposure to LOAELs produced hazard quotients (HQs) greater than one. HQs are defined in Sect. 4.2.3.1.1. (LOAELs are presented in Sect 4.2.2; screening of exposure estimates against LOAELs is presented in Sect. 4.2.3.). Distributions were used to represent contaminant concentrations in sewage sludge from the National Sewage Sludge Survey (Table 4.1), spatial variability in background levels of elements in surface soil (Table 4.28), concentrations in food items based on log-linear bioaccumulation models (Tables 4.6 through 4.9), variability in water content of food types (Table 4.11), variability in body weights of endpoint species (Table 4.10), and variability in food and soil ingestion rates (Table 4.10). The region for which concentrations of elements in background soils were available was sometimes as large as the eastern United States or as small as a single sludge-amended site. Ranges of background values were chosen that were determined to be most applicable to the

Table 4.12. Results of Screening Exposure Assessment for Trowbridge Shrew in Douglas-fir forests

Exposure Scenario	Chemical of Concern	Food Exposure	Soil Exposure	Total Exposure	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.618	0.227	0.846	0.203	2.026	4.17	0.42
	Cadmium	2.100	0.132	2.232	2.866	28.662	0.78	0.08
	Copper	52.703	3.181	55.884	45.2	59.5	1.24	0.94
	Lead	9.221	3.897	13.118	23.77	237.71	0.55	0.06
	Mercury [c]	1.340	0.012	1.352	0.095	0.475	14.23	2.85
	Molybdenum	0.000	0.257	0.257	0.42	4.18	0.61	0.06
	Nickel	30.779	1.251	32.030	118.85	237.71	0.27	0.13
	Selenium	0.198	0.028	0.227	0.594	0.981	0.38	0.23
	Zinc	215.525	14.277	229.802	475.4	950.8	0.48	0.24
	TCDD	9.84E-06	8.51E-08	9.93E-06	3.00e-06	2.97e-05	3.31	0.33
	PCB-1248	3.19E-04	3.22E-04	6.41E-04	0.058	0.58	0.01	0.00
	PCB-1254	7.14E-04	7.19E-04	1.43E-03	0.09	0.9	0.02	0.00
	PCB-1260	2.44E-04	2.46E-04	4.90E-04	0.09	0.9	0.01	0.00
Background Soil [b]	Arsenic	0.551	0.203	0.754	0.203	2.026	3.71	0.37
	Cadmium	1.242	0.078	1.320	2.866	28.662	0.46	0.05
	Copper	28.411	1.715	30.126	45.2	59.5	0.67	0.51
	Lead	8.299	3.508	11.807	23.77	237.71	0.50	0.05
	Mercury [c]	0.426	0.004	0.430	0.095	0.475	4.53	0.91
	Molybdenum	0.000	0.234	0.234	0.42	4.18	0.56	0.06
	Nickel	26.842	1.091	27.933	118.85	237.71	0.24	0.12
	Selenium	0.109	0.016	0.124	0.594	0.981	0.21	0.13
	Zinc	141.206	9.354	150.560	475.4	950.8	0.32	0.16
	TCDD	0.00E+00	0.00E+00	0.00E+00	3.00E-06	2.97E-05	0.00	0.00
	PCB-1248	0.00E+00	0.00E+00	0.00E+00	5.80E-02	5.80E-01	0.00	0.00
	PCB-1254	0.00E+00	0.00E+00	0.00E+00	9.00E-02	9.00E-01	0.00	0.00
	PCB-1260	0.00E+00	0.00E+00	0.00E+00	9.00E-02	9.00E-01	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.2.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.13. Results of Screening Exposure Assessment for Black-tailed Deer in Douglas-fir forests

Exposure Scenario	Chemical of Concern	Food Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.070	0.001	0.071	0.018	0.184	3.95	0.39
	Cadmium	0.120	0.000	0.121	0.26	2.6	0.46	0.05
	Copper	0.559	0.010	0.569	4.1	5.4	0.14	0.11
	Lead	0.512	0.012	0.525	2.16	21.6	0.24	0.02
	Mercury [c]	0.018	0.000	0.018	0.009	0.043	1.95	0.41
	Molybdenum	0.509	0.001	0.510	0.04	0.38	12.76	1.34
	Nickel	0.496	0.004	0.500	10.79	21.57	0.05	0.02
	Selenium	0.024	0.000	0.024	0.054	0.089	0.45	0.27
	Zinc	7.301	0.045	7.346	43.1	86.3	0.17	0.09
	TCDD	0	2.66e-10	2.66e-10	3.00e-07	2.70e-06	0.00	0.00
	PCB-1248	0	1.00e-06	1.00e-06	0.005	0.052	0.00	0.00
	PCB-1254	0	2.24e-06	2.24e-06	0.008	0.082	0.00	0.00
	PCB-1260	0	7.67e-07	7.67e-07	0.008	0.082	0.00	0.00
Background Soil [b]	Arsenic	0.063	0.001	0.063	0.018	0.184	3.52	0.34
	Cadmium	0.071	0.000	0.071	0.26	2.6	0.27	0.03
	Copper	0.301	0.005	0.307	4.1	5.4	0.07	0.06
	Lead	0.461	0.011	0.472	2.16	21.6	0.22	0.02
	Mercury [c]	0.006	0.000	0.006	0.009	0.043	0.62	0.13
	Molybdenum	0.464	0.001	0.465	0.04	0.38	11.62	1.22
	Nickel	0.433	0.003	0.436	10.79	21.57	0.04	0.02
	Selenium	0.013	0.000	0.013	0.054	0.089	0.25	0.15
	Zinc	4.784	0.029	4.813	43.1	86.3	0.11	0.06
	TCDD	0	0	0	3.00e-07	2.70e-06	0.00	0.00
	PCB-1248	0	0	0	0.005	0.052	0.00	0.00
	PCB-1254	0	0	0	0.008	0.082	0.00	0.00
	PCB-1260	0	0	0	0.008	0.082	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.2.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.14. Results of Screening Exposure Assessment for Red Fox in Douglas-fir forests

Exposure Scenario	Chemical of Concern	Mammal Exp	Plant Exp	Invert Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.001	0.007	0.005	0.008	0.021	0.036	0.36	0.60	0.06
	Cadmium	0.175	0.013	0.016	0.005	0.209	0.509	5.09	0.41	0.04
	Copper	1.105	0.059	0.410	0.114	1.688	8	10.6	0.21	0.16
	Lead	0.371	0.054	0.072	0.140	0.637	4.22	42.25	0.15	0.02
	Mercury [c]	0.001	0.002	0.010	0.000	0.013	0.01	0.017	1.35	0.79
	Molybdenum	0.000	0.053	0.000	0.009	0.063	0.07	0.74	0.90	0.08
	Nickel	0.245	0.052	0.239	0.045	0.581	21.12	42.25	0.03	0.01
	Selenium	0.011	0.003	0.002	0.001	0.016	0.106	0.174	0.15	0.09
	Zinc	12.760	0.767	1.675	0.513	15.715	84.5	169	0.19	0.09
	TCDD	6.23E-08	0.00E+00	7.65E-08	3.06E-09	1.42E-07	5.00E-07	5.30E-06	0.28	0.03
	PCB-1248	0.00E+00	0.00E+00	2.48E-06	1.16E-05	1.40E-05	0.01	0.103	0.00	0.00
	PCB-1254	0.00E+00	0.00E+00	5.55E-06	2.58E-05	3.14E-05	0.096	0.474	0.00	0.00
	PCB-1260	0.00E+00	0.00E+00	1.90E-06	8.84E-06	1.07E-05	0.096	0.474	0.00	0.00
Background Soil [b]	Arsenic	0.001	0.007	0.004	0.007	0.019	0.036	0.36	0.53	0.05
	Cadmium	0.103	0.007	0.010	0.003	0.123	0.509	5.09	0.24	0.02
	Copper	0.596	0.032	0.221	0.062	0.910	8	10.6	0.11	0.09
	Lead	0.334	0.048	0.064	0.126	0.573	4.22	42.25	0.14	0.01
	Mercury [c]	0.000	0.001	0.003	0.000	0.004	0.01	0.017	0.43	0.25
	Molybdenum	0.000	0.049	0.000	0.008	0.057	0.07	0.74	0.82	0.08
	Nickel	0.214	0.045	0.209	0.039	0.507	21.12	42.25	0.02	0.01
	Selenium	0.006	0.001	0.001	0.001	0.009	0.106	0.174	0.08	0.05
	Zinc	8.360	0.502	1.097	0.336	10.296	84.5	169	0.12	0.06
	TCDD	0	0	0	0	0	5.00E-07	5.30E-06	0.00	0.00
	PCB-1248	0	0	0	0	0	0.01	0.103	0.00	0.00
	PCB-1254	0	0	0	0	0	0.096	0.474	0.00	0.00
	PCB-1260	0	0	0	0	0	0.096	0.474	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.2.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.15. Results of Screening Exposure Assessment for American Robin in Douglas-fir forests

Exposure Scenario	Chemical of Concern	Plant Exp	Invert Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95%UCL [a]	Arsenic	0.559	0.242	0.072	0.872	5.1	12.8	0.17	0.07
	Cadmium	0.954	0.821	0.042	1.817	1.45	20	1.25	0.09
	Copper	4.428	20.611	1.007	26.046	47	61.7	0.55	0.42
	Lead	4.062	3.606	1.234	8.902	1.13	11.3	7.88	0.79
	Mercury [c]	0.139	0.524	0.004	0.667	0.006	0.064	111.15	10.42
	Molybdenum	4.038	0.000	0.081	4.119	3.5	35.3	1.18	0.12
	Nickel	3.933	12.037	0.396	16.366	77.4	107	0.21	0.15
	Selenium	0.191	0.078	0.009	0.277	0.5	1	0.55	0.28
	Zinc	57.876	84.288	4.519	146.683	14.5	131	10.12	1.12
	TCDD	0.00E+00	3.85E-06	2.70E-08	3.88E-06	1.40E-05	1.40E-04	0.28	0.03
	PCB-1248	0.00E+00	1.25E-04	1.02E-04	2.27E-04	0.18	1.8	0.00	0.00
	PCB-1254	0.00E+00	2.79E-04	2.28E-04	5.07E-04	0.18	1.8	0.00	0.00
	PCB-1260	0.00E+00	9.55E-05	7.79E-05	1.73E-04	0.18	1.8	0.00	0.00
Background Soil [b]	Arsenic	0.498	0.216	0.064	0.778	5.1	12.8	0.15	0.06
	Cadmium	0.564	0.486	0.025	1.075	1.45	20	0.74	0.05
	Copper	2.387	11.111	0.543	14.041	47	61.7	0.30	0.23
	Lead	3.656	3.246	1.110	8.013	1.13	11.3	7.09	0.71
	Mercury [c]	0.044	0.167	0.001	0.212	0.006	0.064	35.38	3.32
	Molybdenum	3.677	0.000	0.074	3.751	3.5	35.3	1.07	0.11
	Nickel	3.430	10.497	0.345	14.273	77.4	107	0.18	0.13
	Selenium	0.105	0.042	0.005	0.152	0.5	1	0.30	0.15
	Zinc	37.919	55.223	2.961	96.103	14.5	131	6.63	0.73
	TCDD	0	0	0	0	1.40E-05	1.40E-04	0.00	0.00
	PCB-1248	0	0	0	0	0.18	1.8	0.00	0.00
	PCB-1254	0	0	0	0	0.18	1.8	0.00	0.00
	PCB-1260	0	0	0	0	0.18	1.8	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.2.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.16. Results of Screening Exposure Assessment for Short-tailed Shrew in Loblolly Pine Plantations

Exposure Scenario	Chemical of Concern	Food Exposure	Soil Exposure	Total Exposure	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.113	0.175	0.288	0.15	1.49	1.92	0.19
	Cadmium	2.901	0.058	2.959	2.12	21.2	1.40	0.14
	Copper	3.642	1.933	5.575	33.4	44	0.17	0.13
	Lead	3.496	1.866	5.362	17.6	175.8	0.30	0.03
	Mercury [c]	0.269	0.011	0.280	0.07	0.35	3.99	0.80
	Molybdenum	0.000	0.096	0.096	0.31	3.09	0.31	0.03
	Nickel	5.273	0.906	6.179	87.91	175.83	0.07	0.04
	Selenium	0.042	0.026	0.068	0.44	0.725	0.15	0.09
	Zinc	86.122	5.431	91.553	351.7	703.3	0.26	0.13
	TCDD	1.83E-06	6.69E-08	1.90E-06	2.00E-06	2.00E-05	0.95	0.09
	PCB-1248	4.95E-03	2.53E-04	5.20E-03	0.043	0.427	0.12	0.01
	PCB-1254	1.11E-02	5.65E-04	1.16E-02	0.067	0.668	0.17	0.02
	PCB-1260	3.79E-03	1.93E-04	3.98E-03	0.067	0.668	0.06	0.01
Background Soil [b]	Arsenic	0.100	0.156	0.256	0.15	1.49	1.71	0.17
	Cadmium	0.781	0.016	0.797	2.12	21.2	0.38	0.04
	Copper	1.470	0.780	2.250	33.4	44	0.07	0.05
	Lead	2.922	1.560	4.482	17.6	175.8	0.25	0.03
	Mercury [c]	0.099	0.004	0.103	0.07	0.35	1.47	0.29
	Molybdenum	0.000	0.078	0.078	0.31	3.09	0.25	0.03
	Nickel	4.541	0.780	5.321	87.91	175.83	0.06	0.03
	Selenium	0.026	0.016	0.041	0.44	0.725	0.09	0.06
	Zinc	24.739	1.560	26.299	351.7	703.3	0.07	0.04
	TCDD	0	0	0	2.00E-06	2.00E-05	0.00	0.00
	PCB-1248	0	0	0	0.043	0.427	0.00	0.00
	PCB-1254	0	0	0	0.067	0.668	0.00	0.00
	PCB-1260	0	0	0	0.067	0.668	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.3.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.17. Results of Screening Exposure Assessment for White-tailed Deer in Loblolly Pine Plantations

Exposure Scenario	Chemical of Concern	Food Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.038	0.001	0.040	0.019	0.19	2.08	0.21
	Cadmium	0.037	0.000	0.038	0.271	2.71	0.14	0.01
	Copper	0.238	0.015	0.254	4.3	5.6	0.06	0.05
	Lead	0.172	0.015	0.187	2.24	22.44	0.08	0.01
	Mercury [c]	0.010	0.000	0.011	0.009	0.085	1.17	0.12
	Molybdenum	0.134	0.001	0.135	0.04	0.39	3.37	0.35
	Nickel	0.252	0.007	0.259	11.22	22.44	0.02	0.01
	Selenium	0.015	0.000	0.015	0.056	0.093	0.28	0.17
	Zinc	1.951	0.043	1.994	44.9	89.8	0.04	0.02
	TCDD	0.00E+00	5.29E-10	5.29E-10	3.00E-07	2.80E-06	0.00	0.00
	PCB-1248	0.00E+00	2.00E-06	2.00E-06	0.005	0.052	0.00	0.00
	PCB-1254	0.00E+00	4.46E-06	4.46E-06	0.008	0.082	0.00	0.00
	PCB-1260	0.00E+00	1.53E-06	1.53E-06	0.008	0.082	0.00	0.00
Background Soil [b]	Arsenic	0.034	0.001	0.035	0.019	0.19	1.85	0.19
	Cadmium	0.010	0.000	0.010	0.271	2.71	0.04	0.00
	Copper	0.096	0.006	0.102	4.3	5.6	0.02	0.02
	Lead	0.144	0.012	0.156	2.24	22.44	0.07	0.01
	Mercury [c]	0.004	0.000	0.004	0.009	0.085	0.43	0.05
	Molybdenum	0.109	0.001	0.109	0.04	0.39	2.73	0.28
	Nickel	0.217	0.006	0.223	11.22	22.44	0.02	0.01
	Selenium	0.009	0.000	0.009	0.056	0.093	0.17	0.10
	Zinc	0.560	0.012	0.573	44.9	89.8	0.01	0.01
	TCDD	0	0	0	3.00E-07	2.80E-06	0.00	0.00
	PCB-1248	0	0	0	0.005	0.052	0.00	0.00
	PCB-1254	0	0	0	0.008	0.082	0.00	0.00
	PCB-1260	0	0	0	0.008	0.082	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.3.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.18. Results of Screening Exposure Assessment for Red Fox in Loblolly Pine Plantations

Exposure Scenario	Chemical of Concern	Mammal Exp	Plant Exp	Invert Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.001	0.006	0.004	0.006	0.017	0.036	0.36	0.46	0.05
	Cadmium	0.077	0.006	0.007	0.002	0.092	0.509	5.09	0.18	0.02
	Copper	0.671	0.036	0.249	0.069	1.025	8	10.6	0.13	0.10
	Lead	0.178	0.026	0.034	0.067	0.305	4.22	42.25	0.07	0.01
	Mercury [c]	0.001	0.002	0.009	0.000	0.011	0.01	0.017	1.14	0.67
	Molybdenum	0.000	0.020	0.000	0.003	0.023	0.07	0.74	0.33	0.03
	Nickel	0.177	0.038	0.173	0.033	0.421	21.12	42.25	0.02	0.01
	Selenium	0.010	0.002	0.001	0.001	0.015	0.106	0.174	0.14	0.08
	Zinc	4.851	0.291	0.637	0.195	5.974	84.5	169	0.07	0.04
	TCDD	4.89E-08	0	6.01E-08	2.40E-09	1.11E-07	5.00E-07	5.30E-06	0.22	0.02
	PCB-1248	0	0	1.95E-06	9.08E-06	1.10E-05	0.01	0.103	0.00	0.00
	PCB-1254	0	0	4.36E-06	2.03E-05	2.47E-05	0.096	0.474	0.00	0.00
	PCB-1260	0	0	1.49E-06	6.94E-06	8.43E-06	0.096	0.474	0.00	0.00
Background Soil [b]	Arsenic	0.001	0.005	0.003	0.006	0.015	0.036	0.36	0.41	0.04
	Cadmium	0.021	0.001	0.002	0.001	0.025	0.509	5.09	0.05	0.00
	Copper	0.271	0.014	0.100	0.028	0.414	8	10.6	0.05	0.04
	Lead	0.148	0.022	0.029	0.056	0.255	4.22	42.25	0.06	0.01
	Mercury [c]	0.000	0.001	0.003	0.000	0.004	0.01	0.017	0.42	0.25
	Molybdenum	0.000	0.016	0.000	0.003	0.019	0.07	0.74	0.27	0.03
	Nickel	0.153	0.032	0.149	0.028	0.362	21.12	42.25	0.02	0.01
	Selenium	0.006	0.001	0.001	0.001	0.009	0.106	0.174	0.08	0.05
	Zinc	1.393	0.084	0.183	0.056	1.716	84.5	169	0.02	0.01
	TCDD	0	0	0	0	0	5.00E-07	5.30E-06	0.00	0.00
	PCB-1248	0	0	0	0	0	0.01	0.103	0.00	0.00
	PCB-1254	0	0	0	0	0	0.096	0.474	0.00	0.00
	PCB-1260	0	0	0	0	0	0.096	0.474	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.3.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.19. Results of Screening Exposure Assessment for American Robin in Loblolly Pine Plantations

Exposure Scenario	Chemical of Concern	Plant Exp	Invert Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.431	0.186	0.055	0.673	5.1	12.8	0.13	0.05
	Cadmium	0.419	0.361	0.018	0.798	1.45	20	0.55	0.04
	Copper	2.689	12.515	0.611	15.815	47	61.7	0.34	0.26
	Lead	1.944	1.726	0.590	4.260	1.13	11.3	3.77	0.38
	Mercury [c]	0.118	0.444	0.003	0.565	0.006	0.064	94.22	8.83
	Molybdenum	1.509	0.000	0.030	1.540	3.5	35.3	0.44	0.04
	Nickel	2.845	8.708	0.287	11.840	77.4	107	0.15	0.11
	Selenium	0.172	0.070	0.008	0.251	0.5	1	0.50	0.25
	Zinc	22.000	32.040	1.718	55.759	14.5	131	3.85	0.43
	TCDD	0	3.02E-06	2.12E-08	3.05E-06	1.40E-05	1.40E-04	0.22	0.02
	PCB-1248	0	9.82E-05	8.00E-05	1.78E-04	0.18	1.8	0.00	0.00
	PCB-1254	0	2.19E-04	1.79E-04	3.98E-04	0.18	1.8	0.00	0.00
	PCB-1260	0	7.51E-05	6.12E-05	1.36E-04	0.18	1.8	0.00	0.00
Background Soil [b]	Arsenic	0.383	0.166	0.049	0.598	5.1	12.8	0.12	0.05
	Cadmium	0.113	0.097	0.005	0.215	1.45	20	0.15	0.01
	Copper	1.085	5.051	0.247	6.382	47	61.7	0.14	0.10
	Lead	1.625	1.443	0.494	3.561	1.13	11.3	3.15	0.32
	Mercury [c]	0.043	0.163	0.001	0.208	0.006	0.064	34.69	3.25
	Molybdenum	1.226	0.000	0.025	1.250	3.5	35.3	0.36	0.04
	Nickel	2.450	7.498	0.247	10.195	77.4	107	0.13	0.10
	Selenium	0.105	0.042	0.005	0.152	0.5	1	0.30	0.15
	Zinc	6.320	9.204	0.494	16.017	14.5	131	1.10	0.12
	TCDD	0	0	0	0	1.40E-05	1.40E-04	0.00	0.00
	PCB-1248	0	0	0	0	0.18	1.8	0.00	0.00
	PCB-1254	0	0	0	0	0.18	1.8	0.00	0.00
	PCB-1260	0	0	0	0	0.18	1.8	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.3.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.20. Results of Screening Exposure Assessment for Short-tailed Shrew in Eastern Deciduous Forest

Exposure Scenario	Chemical of Concern	Food Exposure	Soil Exposure	Total Exposure	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.223	0.347	0.570	0.15	1.49	3.80	0.38
	Cadmium	3.750	0.075	3.824	2.12	21.2	1.80	0.18
	Copper	4.305	2.285	6.590	33.4	44	0.20	0.15
	Lead	7.889	4.212	12.101	17.6	175.8	0.69	0.07
	Mercury [c]	0.339	0.013	0.352	0.07	0.35	5.03	1.01
	Molybdenum	0.000	0.259	0.259	0.31	3.09	0.84	0.08
	Nickel	7.837	1.346	9.183	87.91	175.83	0.10	0.05
	Selenium	0.049	0.030	0.079	0.44	0.725	0.18	0.11
	Zinc	162.009	10.216	172.225	351.7	703.3	0.49	0.24
	TCDD	2.56E-06	9.37E-08	2.66E-06	2.00E-06	2.00E-05	1.33	0.13
	PCB-1248	6.93E-03	3.54E-04	7.29E-03	0.043	0.427	0.17	0.02
	PCB-1254	1.55E-02	7.91E-04	1.63E-02	0.067	0.668	0.24	0.02
	PCB-1260	5.30E-03	2.71E-04	5.57E-03	0.067	0.668	0.08	0.01
Background Soil [b]	Arsenic	0.206	0.320	0.526	0.15	1.49	3.50	0.35
	Cadmium	0.781	0.016	0.797	2.12	21.2	0.38	0.04
	Copper	1.264	0.671	1.935	33.4	44	0.06	0.04
	Lead	7.086	3.783	10.869	17.6	175.8	0.62	0.06
	Mercury [c]	0.101	0.004	0.105	0.07	0.35	1.50	0.30
	Molybdenum	0.000	0.234	0.234	0.31	3.09	0.75	0.08
	Nickel	6.811	1.170	7.981	87.91	175.83	0.09	0.05
	Selenium	0.026	0.016	0.041	0.44	0.725	0.09	0.06
	Zinc	76.073	4.797	80.870	351.7	703.3	0.23	0.11
	TCDD	0	0	0	2.00E-06	2.00E-05	0.00	0.00
	PCB-1248	0	0	0	0.043	0.427	0.00	0.00
	PCB-1254	0	0	0	0.067	0.668	0.00	0.00
	PCB-1260	0	0	0	0.067	0.668	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.4.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.21. Results of Screening Exposure Assessment for White-tailed Deer in Eastern Deciduous Forest

Exposure Scenario	Chemical of Concern	Food Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL[a]	Arsenic	0.076	0.003	0.078	0.019	0.19	4.12	0.41
	Cadmium	0.048	0.001	0.049	0.271	2.71	0.18	0.02
	Copper	0.282	0.018	0.300	4.3	5.6	0.07	0.05
	Lead	0.389	0.033	0.422	2.24	22.44	0.19	0.02
	Mercury[c]	0.013	0.000	0.013	0.009	0.085	1.48	0.16
	Molybdenum	0.361	0.002	0.363	0.04	0.39	9.08	0.93
	Nickel	0.375	0.011	0.386	11.22	22.44	0.03	0.02
	Selenium	0.018	0.000	0.018	0.056	0.093	0.32	0.19
	Zinc	3.671	0.081	3.751	44.9	89.8	0.08	0.04
	TCDD	0	7.40E-10	7.40E-10	3.00E-07	2.80E-06	0.00	0.00
	PCB-1248	0	2.80E-06	2.80E-06	0.005	0.052	0.00	0.00
	PCB-1254	0	6.25E-06	6.25E-06	0.008	0.082	0.00	0.00
	PCB-1260	0	2.14E-06	2.14E-06	0.008	0.082	0.00	0.00
Background Soil[b]	Arsenic	0.070	0.003	0.072	0.019	0.19	3.80	0.38
	Cadmium	0.010	0.000	0.010	0.271	2.71	0.04	0.00
	Copper	0.083	0.005	0.088	4.3	5.6	0.02	0.02
	Lead	0.350	0.030	0.379	2.24	22.44	0.17	0.02
	Mercury[c]	0.004	0.000	0.004	0.009	0.085	0.44	0.05
	Molybdenum	0.326	0.002	0.328	0.04	0.39	8.20	0.84
	Nickel	0.326	0.009	0.335	11.22	22.44	0.03	0.01
	Selenium	0.009	0.000	0.009	0.056	0.093	0.17	0.10
	Zinc	1.724	0.038	1.761	44.9	89.8	0.04	0.02
	TCDD	0	0	0	3.00E-07	2.80E-06	0.00	0.00
	PCB-1248	0	0	0	0.005	0.052	0.00	0.00
	PCB-1254	0	0	0	0.008	0.082	0.00	0.00
	PCB-1260	0	0	0	0.008	0.082	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.4.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.22. Results of Screening Exposure Assessment for Red Fox in Eastern Deciduous Forest

Exposure Scenario	Chemical of Concern	Mammal Exp	Plant Exp	Invert Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.002	0.011	0.007	0.012	0.033	0.036	0.36	0.91	0.09
	Cadmium	0.099	0.007	0.009	0.003	0.118	0.509	5.09	0.23	0.02
	Copper	0.793	0.042	0.294	0.082	1.211	8	10.6	0.15	0.11
	Lead	0.401	0.058	0.077	0.151	0.688	4.22	42.25	0.16	0.02
	Mercury [c]	0.001	0.002	0.011	0.000	0.014	0.01	0.017	1.44	0.85
	Molybdenum	0.000	0.054	0.000	0.009	0.063	0.07	0.74	0.90	0.09
	Nickel	0.264	0.056	0.257	0.048	0.625	21.12	42.25	0.03	0.01
	Selenium	0.012	0.003	0.002	0.001	0.017	0.106	0.174	0.16	0.10
	Zinc	9.125	0.548	1.198	0.367	11.237	84.5	169	0.13	0.07
	TCDD	6.85E-08	0.00E+00	8.41E-08	3.36E-09	1.56E-07	5.00E-07	5.30E-06	0.31	0.03
	PCB-1248	0.00E+00	0.00E+00	2.73E-06	1.27E-05	1.54E-05	0.01	0.103	0.00	0.00
	PCB-1254	0.00E+00	0.00E+00	6.10E-06	2.84E-05	3.45E-05	0.096	0.474	0.00	0.00
	PCB-1260	0.00E+00	0.00E+00	2.09E-06	9.72E-06	1.18E-05	0.096	0.474	0.00	0.00
Background Soil [b]	Arsenic	0.002	0.010	0.007	0.011	0.030	0.036	0.36	0.84	0.08
	Cadmium	0.021	0.001	0.002	0.001	0.025	0.509	5.09	0.05	0.00
	Copper	0.233	0.012	0.086	0.024	0.356	8	10.6	0.04	0.03
	Lead	0.360	0.052	0.070	0.136	0.618	4.22	42.25	0.15	0.01
	Mercury [c]	0.000	0.001	0.003	0.000	0.004	0.01	0.017	0.43	0.25
	Molybdenum	0.000	0.049	0.000	0.008	0.057	0.07	0.74	0.82	0.08
	Nickel	0.229	0.049	0.223	0.042	0.543	21.12	42.25	0.03	0.01
	Selenium	0.006	0.001	0.001	0.001	0.009	0.106	0.174	0.08	0.05
	Zinc	4.285	0.257	0.562	0.172	5.277	84.5	169	0.06	0.03
	TCDD	0	0	0	0	0	5.00E-07	5.30E-06	0.00	0.00
	PCB-1248	0	0	0	0	0	0.01	0.103	0.00	0.00
	PCB-1254	0	0	0	0	0	0.096	0.474	0.00	0.00
	PCB-1260	0	0	0	0	0	0.096	0.474	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.4.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.23. Results of Screening Exposure Assessment for American Robin in Eastern Deciduous Forest

Exposure Scenario	Chemical of Concern	Plant Exp	Invert Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.852	0.369	0.110	1.330	5.1	12.8	0.26	0.10
	Cadmium	0.542	0.466	0.024	1.032	1.45	20	0.71	0.05
	Copper	3.178	14.793	0.723	18.694	47	61.7	0.40	0.30
	Lead	4.387	3.895	1.332	9.614	1.13	11.3	8.51	0.85
	Mercury [c]	0.149	0.560	0.004	0.712	0.006	0.064	118.73	11.13
	Molybdenum	4.074	0.000	0.082	4.156	3.5	35.3	1.19	0.12
	Nickel	4.228	12.941	0.426	17.595	77.4	107	0.23	0.16
	Selenium	0.200	0.081	0.009	0.290	0.5	1	0.58	0.29
	Zinc	41.386	60.273	3.232	104.891	14.5	131	7.23	0.80
	TCDD	0.00E+00	4.23E-06	2.96E-08	4.26E-06	1.40E-05	1.40E-04	0.30	0.03
	PCB-1248	0.00E+00	1.37E-04	1.12E-04	2.49E-04	0.18	1.8	0.00	0.00
	PCB-1254	0.00E+00	3.07E-04	2.50E-04	5.58E-04	0.18	1.8	0.00	0.00
	PCB-1260	0.00E+00	1.05E-04	8.56E-05	1.91E-04	0.18	1.8	0.00	0.00
Background Soil [b]	Arsenic	0.785	0.340	0.101	1.226	5.1	12.8	0.24	0.10
	Cadmium	0.113	0.097	0.005	0.215	1.45	20	0.15	0.01
	Copper	0.933	4.343	0.212	5.489	47	61.7	0.12	0.09
	Lead	3.941	3.498	1.197	8.636	1.13	11.3	7.64	0.76
	Mercury [c]	0.044	0.167	0.001	0.212	0.006	0.064	35.38	3.32
	Molybdenum	3.677	0.000	0.074	3.751	3.5	35.3	1.07	0.11
	Nickel	3.675	11.247	0.370	15.292	77.4	107	0.20	0.14
	Selenium	0.105	0.042	0.005	0.152	0.5	1	0.30	0.15
	Zinc	19.433	28.302	1.518	49.253	14.5	131	3.40	0.38
	TCDD	0	0	0	0	1.40E-05	1.40E-04	0.00	0.00
	PCB-1248	0	0	0	0	0.18	1.8	0.00	0.00
	PCB-1254	0	0	0	0	0.18	1.8	0.00	0.00
	PCB-1260	0	0	0	0	0.18	1.8	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.4.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.24. Results of Screening Exposure Assessment for Desert Shrew in Semiarid Rangeland

Exposure Scenario	Chemical of Concern	Food Exposure	Soil Exposure	Total Exposure	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.267	0.188	0.456	0.195	1.95	2.34	0.23
	Cadmium	0.699	0.084	0.783	2.76	27.6	0.28	0.03
	Copper	13.042	1.509	14.551	43.6	57.3	0.33	0.25
	Lead	1.550	1.256	2.806	22.9	229	0.12	0.01
	Mercury [c]	0.602	0.011	0.612	0.092	0.46	6.66	1.33
	Molybdenum	0.000	0.025	0.025	0.4	4	0.06	0.01
	Nickel	12.942	1.009	13.951	114.6	229.1	0.12	0.06
	Selenium	0.316	0.087	0.403	0.57	0.95	0.71	0.42
	Zinc	54.190	6.882	61.072	458	917	0.13	0.07
	TCDD	3.98E-06	6.60E-08	4.05E-06	3.00E-06	2.80E-05	1.35	0.14
	PCB-1248	1.29E-04	2.49E-04	3.79E-04	0.056	0.557	0.01	0.00
	PCB-1254	2.89E-04	5.57E-04	8.46E-04	0.087	0.87	0.01	0.00
	PCB-1260	9.88E-05	1.91E-04	2.90E-04	0.087	0.87	0.00	0.00
Background Soil [b]	Arsenic	0.240	0.169	0.409	0.195	1.95	2.10	0.21
	Cadmium	0.352	0.042	0.394	2.76	27.6	0.14	0.01
	Copper	3.217	0.372	3.590	43.6	57.3	0.08	0.06
	Lead	1.177	0.954	2.131	22.9	229	0.09	0.01
	Mercury [c]	0.232	0.004	0.237	0.092	0.46	2.57	0.51
	Molybdenum	0.000	0.008	0.008	0.4	4	0.02	0.00
	Nickel	11.350	0.885	12.234	114.6	229.1	0.11	0.05
	Selenium	0.280	0.077	0.357	0.57	0.95	0.63	0.38
	Zinc	24.132	3.065	27.197	458	917	0.06	0.03
	TCDD	0	0	0	3.00E-06	2.80E-05	0.00	0.00
	PCB-1248	0	0	0	0.056	0.557	0.00	0.00
	PCB-1254	0	0	0	0.087	0.87	0.00	0.00
	PCB-1260	0	0	0	0.087	0.87	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.5.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.25. Results of Screening Exposure Assessment for Mule Deer in Semiarid Rangeland

Exposure Scenario	Chemical of Concern	Food Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.059	0.001	0.060	0.018	0.184	3.32	0.32
	Cadmium	0.078	0.000	0.078	0.26	2.6	0.30	0.03
	Copper	0.269	0.005	0.273	4.1	5.4	0.07	0.05
	Lead	0.167	0.004	0.171	2.16	21.6	0.08	0.01
	Mercury [c]	0.015	0.000	0.015	0.009	0.043	1.71	0.36
	Molybdenum	0.051	0.000	0.051	0.04	0.38	1.28	0.14
	Nickel	0.405	0.003	0.408	10.79	21.57	0.04	0.02
	Selenium	0.075	0.000	0.075	0.054	0.089	1.39	0.84
	Zinc	3.566	0.022	3.588	43.1	86.3	0.08	0.04
	TCDD	0.00E+00	2.09E-10	2.09E-10	3.00E-07	2.70E-06	0.00	0.00
	PCB-1248	0.00E+00	7.88E-07	7.88E-07	0.005	0.052	0.00	0.00
	PCB-1254	0.00E+00	1.76E-06	1.76E-06	0.008	0.082	0.00	0.00
	PCB-1260	0.00E+00	6.03E-07	6.03E-07	0.008	0.082	0.00	0.00
Background Soil [b]	Arsenic	0.053	0.001	0.054	0.018	0.184	2.98	0.29
	Cadmium	0.039	0.000	0.039	0.26	2.6	0.15	0.02
	Copper	0.066	0.001	0.067	4.1	5.4	0.02	0.01
	Lead	0.127	0.003	0.130	2.16	21.6	0.06	0.01
	Mercury [c]	0.006	0.000	0.006	0.009	0.043	0.66	0.14
	Molybdenum	0.015	0.000	0.015	0.04	0.38	0.39	0.04
	Nickel	0.355	0.003	0.358	10.79	21.57	0.03	0.02
	Selenium	0.066	0.000	0.066	0.054	0.089	1.23	0.74
	Zinc	1.588	0.010	1.598	43.1	86.3	0.04	0.02
	TCDD	0	0	0	3.00E-07	2.70E-06	0.00	0.00
	PCB-1248	0	0	0	0.005	0.052	0.00	0.00
	PCB-1254	0	0	0	0.008	0.082	0.00	0.00
	PCB-1260	0	0	0	0.008	0.082	0.00	0.00

a Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

b Soil concentration based on mean background concentrations obtained from the literature. See Table 4.5.

c Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.26. Results of Screening Exposure Assessment for Kit Fox in Semiarid Rangeland

Expsoure Scenario	Chemical of Concern	mammal Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.002	0.007	0.009	0.044	0.436	0.20	0.02
	Cadmium	0.216	0.003	0.219	0.616	6.16	0.36	0.04
	Copper	1.015	0.056	1.071	9.7	12.8	0.11	0.08
	Lead	0.232	0.047	0.278	5.11	51.1	0.05	0.01
	Mercury [c]	0.001	0.000	0.002	0.012	0.021	0.14	0.08
	Molybdenum	0.000	0.001	0.001	0.09	0.9	0.01	0.00
	Nickel	0.383	0.037	0.420	25.56	51.12	0.02	0.01
	Selenium	0.066	0.003	0.070	0.128	0.211	0.54	0.33
	Zinc	11.909	0.256	12.164	102.2	204.5	0.12	0.06
	TCDD	9.35E-08	2.45E-09	9.60E-08	5.00E-07	5.30E-06	0.19	0.02
	PCB-1248	0.00E+00	9.26E-06	9.26E-06	0.01	0.124	0.00	0.00
	PCB-1254	0.00E+00	2.07E-05	2.07E-05	0.116	0.573	0.00	0.00
	PCB-1260	0.00E+00	7.08E-06	7.08E-06	0.116	0.573	0.00	0.00
Background Soil [b]	Arsenic	0.002	0.006	0.008	0.044	0.436	0.18	0.02
	Cadmium	0.109	0.002	0.110	0.616	6.16	0.18	0.02
	Copper	0.250	0.014	0.264	9.7	12.8	0.03	0.02
	Lead	0.176	0.035	0.211	5.11	51.1	0.04	0.00
	Mercury [c]	0.001	0.000	0.001	0.012	0.021	0.06	0.03
	Molybdenum	0.000	0.000	0.000	0.09	0.9	0.00	0.00
	Nickel	0.336	0.033	0.368	25.56	51.12	0.01	0.01
	Selenium	0.059	0.003	0.062	0.128	0.211	0.48	0.29
	Zinc	5.303	0.114	5.417	102.2	204.5	0.05	0.03
	TCDD	0	0	0	5.00E-07	5.30E-06	0.00	0.00
	PCB-1248	0	0	0	0.01	0.124	0.00	0.00
	PCB-1254	0	0	0	0.116	0.573	0.00	0.00
	PCB-1260	0	0	0	0.116	0.573	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.5.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.27. Results of Screening Exposure Assessment for Western Meadowlark in Semiarid Rangeland

Exposure Scenario	Chemical of Concern	plant Exp	invert Exp	Soil Exp	Total Exp	NOAEL	LOAEL	NOAEL HQ	LOAEL HQ
NSSS 95% UCL [a]	Arsenic	0.020	0.009	0.063	0.092	5.1	12.8	0.02	0.01
	Cadmium	0.027	0.023	0.028	0.078	1.45	20	0.05	0.00
	Copper	0.093	0.426	0.507	1.025	47	61.7	0.02	0.02
	Lead	0.058	0.051	0.421	0.530	1.13	11.3	0.47	0.05
	Mercury [c]	0.005	0.020	0.004	0.029	0.006	0.064	4.76	0.45
	Molybdenum	0.018	0.000	0.009	0.026	3.5	35.3	0.01	0.00
	Nickel	0.140	0.423	0.339	0.901	77.4	107	0.01	0.01
	Selenium	0.026	0.010	0.029	0.065	0.5	1	0.13	0.07
	Zinc	1.232	1.769	2.310	5.311	14.5	131	0.37	0.04
	TCDD	0.00E+00	1.30E-07	2.22E-08	1.52E-07	1.40E-05	1.40E-04	0.01	0.00
	PCB-1248	0.00E+00	4.22E-06	8.37E-05	8.79E-05	0.18	1.8	0.00	0.00
	PCB-1254	0.00E+00	9.43E-06	1.87E-04	1.97E-04	0.18	1.8	0.00	0.00
	PCB-1260	0.00E+00	3.23E-06	6.40E-05	6.72E-05	0.18	1.8	0.00	0.00
Background Soil [b]	Arsenic	0.018	0.008	0.057	0.083	5.1	12.8	0.02	0.01
	Cadmium	0.014	0.011	0.014	0.039	1.45	20	0.03	0.00
	Copper	0.023	0.105	0.125	0.253	47	61.7	0.01	0.00
	Lead	0.044	0.038	0.320	0.403	1.13	11.3	0.36	0.04
	Mercury [c]	0.002	0.008	0.001	0.011	0.006	0.064	1.84	0.17
	Molybdenum	0.005	0.000	0.003	0.008	3.5	35.3	0.00	0.00
	Nickel	0.123	0.371	0.297	0.790	77.4	107	0.01	0.01
	Selenium	0.023	0.009	0.026	0.058	0.5	1	0.12	0.06
	Zinc	0.549	0.788	1.029	2.365	14.5	131	0.16	0.02
	TCDD	0	0	0	0	1.40E-05	1.40E-04	0.00	0.00
	PCB-1248	0	0	0	0	0.18	1.8	0.00	0.00
	PCB-1254	0	0	0	0	0.18	1.8	0.00	0.00
	PCB-1260	0	0	0	0	0.18	1.8	0.00	0.00

[a] Soil concentration based on Estimated Chemical Concentration in Top 15 cm of soil, following a 1x application of 40 Mg/ha containing 95% UCL concentrations from the NSSS

[b] Soil concentration based on mean background concentrations obtained from the literature. See Table 4.5.

[c] Assumed to be 100 % methyl mercury. NOAEL and LOAEL values are for methyl mercury

Table 4.28. Summary of distribution parameters for chemical concentrations in background soils.

Ecosystem	Chemical	Relevant Endpoint	Concentrations (mg/kg dry wt) and distribution	Reference	Comments
Douglas-Fir Forest	Mercury	wildlife	min=0.009 med=0.038 max=0.185 Triangular	WADOE 1994	Values from samples from western Washington
	Molybdenum	wildlife	0.085±2.17 Lognormal	Schacklette and Boergen 1974	Geometric mean and deviation of values from samples from western United States
	Zinc	wildlife, plant	min=21.1 med=44.5 max=116 Triangular	WADOE 1994	Data from samples from western Washington
Eastern Deciduous Forest and Loblolly Pine Plantation	Mercury	wildlife	min=0.01 geometric mean=0.081 max=3.4 Triangular	Schacklette and Boergen 1974	Values from samples from eastern United States
	Zinc	plant	min=1.5 median=51.8 max=400 Triangular	White et al. 1997	Data from USGS (minus 3 outliers) and USDA for coterminous U. S.
Semiarid Rangeland	Mercury	wildlife	0.0558±0.0106 Normal	Harris-Pierce 1994	Arithmetic mean of values at a sludge treated site in Colorado

	Zinc	plant	min=34.6 geom mean (assumed)=40 max=66	Harris-Pierce 1994, Loftin 1996, Cosby-Horton 1997	Values for rangeland (minimum from Colorado, maximum from western Texas, mean from New Mexico)

particular ecosystem of concern. Triangular distributions were applied to parameters for which data consisted of minimum, median (or mean), and maximum values (e.g., contaminant concentrations in sewage sludge and background soils and most bodyweights and food ingestion rates). Normal distributions were applied to parameters for log-linear bioaccumulation models and water content in food types.

Monte Carlo simulations were performed using the @Risk software (Palisades Corp. 1994). Samples from each distribution were selected using latin hypercube sampling. The number of iterations, or recalculations, of each exposure simulation was determined by the convergence criteria set in the software. Under these criteria, iterations are performed until the between-iteration percent changes in the percentiles, mean, and standard deviation are below 1.5% (i.e., the percentile, mean, and standard deviation for the latest iteration is less than 1.5% different from those from the previous iteration). Using these convergence criteria, from 200 to 1000 model iterations were performed for each exposure estimate. Monte Carlo estimates of contaminant exposures are presented in Table 4.29.

Table 4.29. Summary of results of Monte Carlo simulation of exposure to contaminants following a one time application of sludge containing the 95% UCL concentration from the 1988 NSSS at a rate of 40 Mg/ha. Results presented only for those contaminants and endpoints where screening estimates of exposure exceeded LOAELs.

			Exposure Estimate (mg/kg/d)			
Ecosystem	**Endpoint Species**	**chemical**	**Mean**	**Standard Deviation**	**80th Percentile**	**Maximum**
Douglas-fir Forest	Trowbridge Shrew	mercury	0.114	0.057	0.156	0.301
	Black-tailed Deer	molybdenum	0.02	0.009	0.027	0.105
	American Robin	mercury	0.083	0.027	0.105	0.198
		zinc	106	53.44	141.72	411.88
Eastern Deciduous Forest and Loblolly Pine Plantation	Short-tailed Shrew	mercury	0.071	0.028	0.096	0.157
	American Robin	mercury	0.136	0.0552	0.179	0.307
Semiarid Rangeland	Desert Shrew	mercury	0.035	0.0089	0.042	0.071

4.2.2 Effects Assessment for Wildlife

Traditionally, effects assessments in ecological risk assessments may consist of three types of information: single chemical toxicity data obtained from the literature, toxicity tests

using field-collected media, and biological survey data from field applications. To evaluate potential effects of sludge application on wildlife, the primary available data consist of single chemical toxicity data and biological survey data. In addition, toxicity test results are available for shrews.

4.2.2.1 Single chemical toxicity data

Toxicity values for oral exposures

Single chemical toxicity data consist of no observed adverse effects levels (NOAELs) and lowest observed adverse effects levels (LOAELs) from toxicity studies reported in the literature. NOAELs and LOAELs for wildlife endpoint species were derived according to the methods outlined in Sample et al. (1996b) and briefly summarized below. So that toxicity values would be most relevant to potential long-term population-level effects, chronic studies of reproductive effects following oral exposures were selected for derivation of the wildlife NOAELs and LOAELs, wherever possible.

NOAELs and LOAELs are daily dose levels normalized to the body weight of the test animals (e.g., milligrams of chemical per kilogram body weight per day). The presentation of toxicity data on a mg/kg/day basis allows comparisons across tests and across species with appropriate consideration for differences in body size. Studies have shown that numerous physiological functions such as metabolic rates, as well as responses to toxic chemicals, are a function of body size. Smaller animals have higher metabolic rates and usually are more resistant to toxic chemicals because of more rapid rates of detoxification. (However, this generalization may not be true if the toxic effects of the compound are produced primarily by a metabolite). For mammals, it has been shown that the relationship between size and metabolic rate is best expressed in terms of body weight (bw) raised to the 3/4 power ($bw^{3/4}$) (Travis and White 1988, Travis et al. 1990, EPA 1992a). If the dose (d) has been calculated in terms of unit body weight (i.e., mg/kg/d), then the metabolic rate-based dose (D) equates to:

$$D = \frac{d \; x \; bw}{bw^{3/4}} = d \; x \; bw^{1/4}.$$

The assumption is that the metabolic rate-based dose (Eq. 1) for species "a" and "b" would be equivalent:

$$d_a \; x \; bw_a^{1/4} = d_b \; x \; bw_b^{1/4}.$$

Therefore, knowing the body weights of two species and the dose (d_b) producing a given effect in species "b," the dose (d_a) producing the same effect in species "a" can be determined:

$$d_a = d_b \; x \; \frac{bw_b^{1/4}}{bw_a^{1/4}} = d_b \; x \left(\frac{bw_b}{bw_a} \right)^{1/4}.$$

If a NOAEL (or LOAEL) is available for a mammalian test species ($NOAEL_t$), then the equivalent NOAEL (or LOAEL) for a mammalian wildlife species ($NOAEL_w$) can be calculated by using the adjustment factor for differences in body size:

$$NOAEL_w = NOAEL_t \left(\frac{bw_t}{bw_w} \right)^{1/4}.$$

Recent research suggests that physiological scaling factors developed for mammals may not be appropriate for interspecies extrapolation among birds. Mineau et al. (1996) developed body weight-based scaling factors for birds using LC_{50} data for 37 pesticides. Scaling factors ranged from 0.63 to 1.55 with a mean of 1.15. However, scaling factors for the majority of the chemicals evaluated (29 of 37) were not significantly different from 1. A scaling factor of 1 was therefore considered most appropriate for interspecies extrapolation among birds. If the dose (d) itself has been calculated in terms of unit body weight (i.e., mg/kg), then the extrapolated dose (D) equates to:

$$D = \frac{d \ x \ bw}{bw^1} = d \ x \ bw^0.$$

For birds, if a NOAEL was available for an avian test species ($NOAEL_t$), the equivalent NOAEL for an avian wildlife species ($NOAEL_w$) would be calculated by using the 'adjustment factor' for differences in body size:

$$NOAEL_w = NOAEL_t \left(\frac{bw_t}{bw_w} \right)^0 = NOAEL_t \ (1) = NOAEL_t$$

Because the 'adjustment factor' equals 1, NOAELs and LOAELs for birds are assumed to be equivalent for all species, regardless of body weight.

In cases where a NOAEL for a specific chemical was not available, but a LOAEL had been determined experimentally or where the NOAEL was from a subchronic study, the chronic NOAEL was estimated. EPA (1993b) suggests the use of uncertainty factors of 1 to 10 for subchronic to chronic NOAEL and LOAEL to NOAEL estimation. Because no data were available to suggest the use of lower values, uncertainty factors of 10 were used in all instances where such estimates were required (Sample et al. 1996b)

NOAELs and LOAELs were derived for all nine wildlife endpoint species considered. Experimental information and assumptions used to derive avian and mammalian NOAELs and LOAELs for each contaminant are outlined Sample et al. (1996b). Estimated NOAELs and LOAELs for all wildlife endpoints are listed in Table 4.30.

Toxicity values for body burdens

In addition to oral exposure, toxicity may be evaluated using concentrations in target tissues. Target organ concentrations associated with adverse effects in birds and mammals were

Table 4.30. Summary of estimated NOAELs and LOAELs for wildlife endpoint species. Descriptions of studies and derivation of NOAELs and LOAELs for test species are located in Section 4.2.2.1.2.

Endpoint Species	Chemical	Form	Test Species	TestSpecies NOAEL (mg/kg/day)	Test Species LOAEL (mg/kg/day)	Estimated Wildlife NOAEL (mg/kg/day)	Estimated Wildlife LOAEL (mg/kg/day)
American Robin	Arsenic	sodium arsenite	mallard duck	5.14	12.84	5.1	12.8
	Cadmium	cadmium chloride	mallard duck	1.45	20	1.45	20.00
	Copper	copper oxide	1 day old chicks	47	61.7	47.0	61.7
	Lead	lead acetate	Japanese quail	1.13	11.3	1.13	11.30
	Mercury	mercuric chloride	Japanese Quail	0.45	0.9	0.45	0.90
	Mercury	Methyl Mercury Dicyandiamide	mallard duck	0.0064	0.064	0.006	0.064
	Molybdenum	sodium molybdate (MoO4)	chicken	3.5	35.3	3.50	35.30
	Nickel	nickel sulfate	mallard duckling	77.4	107	77.40	107.00
	Selenium	sodium selenite	mallard duck	0.5	1	0.500	1.000
	Zinc	zinc sulfate	white leghorn hen	14.5	131	14.5	131.0
	2,3,7,8-TCDD	n/a	ring-necked pheasant	0.000014	0.00014	0.0000140	0.0001400
	Aroclor 1248	Aroclor 1254 values used					
	Aroclor 1254	n/a	ring-necked pheasant	0.18	1.8	0.180	1.800
	Aroclor 1260	Aroclor 1254 values used					
Western Meadowlark	Arsenic	sodium arsenite	mallard duck	5.14	12.84	5.1	12.8
	Cadmium	cadmium chloride	mallard duck	1.45	20	1.45	20.00
	Copper	copper oxide	1 day old chicks	47	61.7	47.0	61.7
	Lead	lead acetate	Japanese quail	1.13	11.3	1.13	11.30
	Mercury	mercuric chloride	Japanese Quail	0.45	0.9	0.45	0.90
	Mercury	Methyl Mercury Dicyandiamide	mallard duck	0.0064	0.064	0.006	0.064
	Molybdenum	sodium molybdate (MoO4)	chicken	3.5	35.3	3.50	35.30
	Nickel	nickel sulfate	mallard duckling	77.4	107	77.40	107.00
	Selenium	sodium selenite	mallard duck	0.5	1	0.500	1.000
	Zinc	zinc sulfate	white leghorn hen	14.5	131	14.5	131.0

Endpoint Species	Chemical	Form	Test Species	TestSpecies NOAEL (mg/kg/day)	Test Species LOAEL (mg/kg/day)	Estimated Wildlife NOAEL (mg/kg/day)	Estimated Wildlife LOAEL (mg/kg/day)
	2,3,7,8-TCDD	n/a	ring-necked pheasant	0.000014	0.00014	0.0000140	0.0001400
	Aroclor 1248	Aroclor 1254 values used					
	Aroclor 1254	n/a	ring-necked pheasant	0.18	1.8	0.180	1.800
	Aroclor 1260	Aroclor 1254 values used					
Short-tailed Shrew	Arsenic	Arsenite	mouse	0.126	1.26	0.150	1.498
	Cadmium	cadmium chloride	rat	1	10	2.120	21.200
	Copper	copper sulfate	mink	11.7	15.4	33.4	44.0
	Lead	lead acetate	rat	8	80	17.58	175.83
	Mercury	mercuric chloride	mink	1		2.86	
	Mercury	Methyl Mercury Chloride	rat	0.032	0.16	0.070	0.352
	Molybdenum	MoO4	mouse	0.26	2.6	0.31	3.09
	Nickel	nickel sulfate hexahydrate	rat	40	80	87.91	175.83
	Selenium	Selenate (SeO4)	rat	0.2	0.33	0.440	0.725
	Zinc	zinc oxide	rat	160	320	351.7	703.3
	2,3,7,8-TCDD	n/a	rat	0.000001	0.00001	0.0000022	0.0000220
	Aroclor 1248	n/a	Rhesus monkey	0.01	0.1	0.043	0.427
	Aroclor 1254	n/a	oldfield mouse	0.068	0.68	0.067	0.668
	Aroclor 1260	Aroclor 1254 values used					
Trowbridge Shrew	Arsenic	Arsenite	mouse	0.126	1.26	0.203	2.026

Endpoint Species	Chemical	Form	Test Species	TestSpecies NOAEL (mg/kg/day)	Test Species LOAEL (mg/kg/day)	Estimated Wildlife NOAEL (mg/kg/day)	Estimated Wildlife LOAEL (mg/kg/day)
	Cadmium	cadmium chloride	rat	1	10	2.866	28.662
	Copper	copper sulfate	mink	11.7	15.4	45.2	59.5
	Lead	lead acetate	rat	8	80	23.77	237.71
	Mercury	mercuric chloride	mink	1		3.86	
	Mercury	Methyl Mercury Chloride	rat	0.032	0.16	0.095	0.475
	Molybdenum	MoO4	mouse	0.26	2.6	0.42	4.18
	Nickel	nickel sulfate hexahydrate	rat	40	80	118.85	237.71
	Selenium	Selenate (SeO4)	rat	0.2	0.33	0.594	0.981
	Zinc	zinc oxide	rat	160	320	475.4	950.8
	2,3,7,8-TCDD	n/a	rat	0.000001	0.00001	0.0000030	0.0000297
	Aroclor 1248	n/a	Rhesus monkey	0.01	0.1	0.058	0.578
	Aroclor 1254	n/a	oldfield mouse	0.068	0.68	0.090	0.904
	Aroclor 1260	Aroclor 1254 values used					
Desert Shrew	Arsenic	Arsenite	mouse	0.126	1.26	0.195	1.953
	Cadmium	cadmium chloride	rat	1	10	2.763	27.629
	Copper	copper sulfate	mink	11.7	15.4	43.6	57.3
	Lead	lead acetate	rat	8	80	22.91	229.14
	Mercury	mercuric chloride	mink	1		3.72	
	Mercury	Methyl Mercury Chloride	rat	0.032	0.16	0.092	0.458
	Molybdenum	MoO4	mouse	0.26	2.6	0.40	4.03

Endpoint Species	Chemical	Form	Test Species	TestSpecies NOAEL (mg/kg/day)	Test Species LOAEL (mg/kg/day)	Estimated Wildlife NOAEL (mg/kg/day)	Estimated Wildlife LOAEL (mg/kg/day)
	Nickel	nickel sulfate hexahydrate	rat	40	80	114.57	229.14
	Selenium	Selenate (SeO4)	rat	0.2	0.33	0.573	0.945
	Zinc	zinc oxide	rat	160	320	458.3	916.6
	2,3,7,8-TCDD	n/a	rat	0.000001	0.00001	0.0000029	0.0000286
	Aroclor 1248	n/a	Rhesus monkey	0.01	0.1	0.056	0.557
	Aroclor 1254	n/a	oldfield mouse	0.068	0.68	0.087	0.871
	Aroclor 1260	Aroclor 1254 values used					
Red Fox	Arsenic	Arsenite	mouse	0.126	1.26	0.036	0.360
	Cadmium	cadmium chloride	rat	1	10	0.509	5.094
	Copper	copper sulfate	mink	11.7	15.4	8.0	10.6
	Lead	lead acetate	rat	8	80	4.22	42.25
	Mercury	mercuric chloride	mink	1		0.69	
	Mercury	Methyl Mercury Chloride	mink	0.015	0.025	0.010	0.017
	Molybdenum	MoO4	mouse	0.26	2.6	0.07	0.74
	Nickel	nickel sulfate hexahydrate	rat	40	80	21.12	42.25
	Selenium	Selenate (SeO4)	rat	0.2	0.33	0.106	0.174
	Zinc	zinc oxide	rat	160	320	84.5	169.0
	2,3,7,8-TCDD	n/a	rat	0.000001	0.00001	0.0000005	0.0000053
	Aroclor 1248	n/a	Rhesus monkey	0.01	0.1	0.010	0.103
	Aroclor 1254	n/a	mink	0.14	0.69	0.096	0.474

Endpoint Species	Chemical	Form	Test Species	TestSpecies NOAEL (mg/kg/day)	Test Species LOAEL (mg/kg/day)	Estimated Wildlife NOAEL (mg/kg/day)	Estimated Wildlife LOAEL (mg/kg/day)
	Aroclor 1260	Aroclor 1254 values used					
Kit Fox	Arsenic	Arsenite	mouse	0.126	1.26	0.044	0.436
	Cadmium	cadmium chloride	rat	1	10	0.616	6.163
	Copper	copper sulfate	mink	11.7	15.4	9.7	12.8
	Lead	lead acetate	rat	8	80	5.11	51.12
	Mercury	mercuric chloride	mink	1		0.83	
	Mercury	Methyl Mercury Chloride	mink	0.015	0.025	0.012	0.021
	Molybdenum	MoO4	mouse	0.26	2.6	0.09	0.90
	Nickel	nickel sulfate hexahydrate	rat	40	80	25.56	51.12
	Selenium	Selenate (SeO4)	rat	0.2	0.33	0.128	0.211
	Zinc	zinc oxide	rat	160	320	102.2	204.5
	2,3,7,8-TCDD	n/a	rat	0.000001	0.00001	0.0000006	0.0000064
	Aroclor 1248	n/a	Rhesus monkey	0.01	0.1	0.012	0.124
	Aroclor 1254	n/a	mink	0.14	0.69	0.116	0.573
	Aroclor 1260	Aroclor 1254 values used					
Whitetail Deer	Arsenic	Arsenite	mouse	0.126	1.26	0.019	0.191
	Cadmium	cadmium chloride	rat	1	10	0.271	2.706
	Copper	copper sulfate	mink	11.7	15.4	4.3	5.6
	Lead	lead acetate	rat	8	80	2.24	22.44
	Mercury	mercuric chloride	mink	1		0.36	

Endpoint Species	Chemical	Form	Test Species	TestSpecies NOAEL (mg/kg/day)	Test Species LOAEL (mg/kg/day)	Estimated Wildlife NOAEL (mg/kg/day)	Estimated Wildlife LOAEL (mg/kg/day)
	Mercury	Methyl Mercury Chloride	rat	0.032	0.16	0.009	0.045
	Molybdenum	MoO4	mouse	0.26	2.6	0.04	0.39
	Nickel	nickel sulfate hexahydrate	rat	40	80	11.22	22.44
	Selenium	Selenate (SeO4)	rat	0.2	0.33	0.056	0.093
	Zinc	zinc oxide	rat	160	320	44.9	89.8
	2,3,7,8-TCDD	n/a	rat	0.000001	0.00001	0.0000003	0.0000028
	Aroclor 1248	n/a	Rhesus monkey	0.01	0.1	0.005	0.055
	Aroclor 1254	n/a	oldfield mouse	0.068	0.68	0.009	0.085
	Aroclor 1260	Aroclor 1254 values used					
Mule/Blacktail Deer	Arsenic	Arsenite	mouse	0.126	1.26	0.018	0.184
	Cadmium	cadmium chloride	rat	1	10	0.260	2.601
	Copper	copper sulfate	mink	11.7	15.4	4.1	5.4
	Lead	lead acetate	rat	8	80	2.16	21.57
	Mercury	mercuric chloride	mink	1		0.35	
	Mercury	Methyl Mercury Chloride	rat	0.032	0.16	0.009	0.043
	Molybdenum	MoO4	mouse	0.26	2.6	0.04	0.38
	Nickel	nickel sulfate hexahydrate	rat	40	80	10.79	21.57
	Selenium	Selenate (SeO4)	rat	0.2	0.33	0.054	0.089
	Zinc	zinc oxide	rat	160	320	43.1	86.3
	2,3,7,8-TCDD	n/a	rat	0.000001	0.00001	0.0000003	0.0000027

Endpoint Species	Chemical	Form	Test Species	TestSpecies NOAEL (mg/kg/day)	Test Species LOAEL (mg/kg/day)	Estimated Wildlife NOAEL (mg/kg/day)	Estimated Wildlife LOAEL (mg/kg/day)
	Aroclor 1248	n/a	Rhesus monkey	0.01	0.1	0.005	0.052
	Aroclor 1254	n/a	oldfield mouse	0.068	0.68	0.008	0.082
	Aroclor 1260	Aroclor 1254 values used					

summarized for the most commonly analyzed contaminants: cadmium, copper, lead, and zinc. These values are presented in Table 4.31.

Cadmium. Leach et al. (1979) observed a 50% reduction in egg production among chickens consuming a diet containing 48 mg/kg cadmium. Cadmium concentrations in the livers and kidneys of these birds were 541 mg/kg and 204 mg/kg, dry weight, respectively. Among mammals, Itokawa et al. (1974) observed severe renal damage in rats with cadmium concentrations in kidneys of 123 mg/kg dry weight. In a recent review of cadmium literature, Cooke and Johnson (1996) suggest that the critical threshold concentration for adverse effects in kidneys is 350 mg/kg dry weight (100 mg/kg wet weight). This concentration has been associated with proteinuria and tubular dysfunction in rodents.

Lead. Getz et al. (1977) sampled birds from urban and rural areas in Illinois. While lead content of liver and kidney of robins from urban areas was significantly greater than that observed in robins from rural areas (Table 4.31), adverse effects were not suspected.

Beyer et al. (1988) dosed six species of birds (red-winged blackbird, brown-headed cowbird, common grackle, mallard, northern bobwhite, and eastern screech owls) with progressively higher doses of lead acetate until 50% of the birds in each group had died. Lead body burdens of all lead-exposed birds were then analyzed. Passerine species experienced mortality earlier than did non-passerine species. Red-winged blackbirds were most sensitive to lead; 50% mortality was observed at 88 mg/kg and 52.6 mg/kg in kidney and liver, respectively. Bobwhite quail were least sensitive, with 50% mortality observed at 74 mg/kg and 340 mg/kg in kidney and liver, respectively. Liver and kidney concentrations of lead are presented Table 4.31.

Among mammals, Ma (1996) reports that lead toxicity is not noted below a tissue level of 5 mg/kg dry weight liver or 10 mg/kg dry weight kidney.

Copper. Liver is the primary organ of copper metabolism (Buck 1978). In sheep, copper concentrations in the liver of 30-150 mg/kg dry weight are considered normal. Copper concentrations in excess of 450 mg/kg dry weight liver may be toxic (Buck 1978). The author noted, however, that sheep store copper more readily in their livers than do other species.

Mehring et al. (1960) evaluated the tolerance of growing chickens to copper. While consumption of 570 mg/kg in the diet had no effect on growth of chicks, 749 mg/kg in the diet reduced growth by over 30% and produced 15% mortality. The liver concentration associated with the NOAEL was 184 mg/kg dry weight; the concentration associated with the LOAEL was 595 mg/kg dry weight.

Zinc. There is little information concerning zinc body burdens associated with toxicity. Reece et al. (1986) analyzed zinc content in livers of peach-faced love birds (*Agapornis roseicollis*) that died within 5 weeks of being placed in new, galvinized wire cages. Zinc content in liver of these birds ranged from 75-156 mg/kg dry weight. Livers of unexposed birds that had died of trauma contained 12-33 mg zinc /kg dry weight.

Gasaway and Buss (1972) fed mallards diets containing 3000 to 12000 mg/kg zinc carbonate for 60 days. Severe mortality was observed among all dose levels. Only two birds in the 3000

Table 4.31. Literature-derived target organ concentrations associated with toxicity					
Analyte	**Taxa**	**Organ Concentration (mg/kg dry)**		**Comments**	**Reference**
		Liver	**Kidney**		
Cadmium	Chicken	204	541	egg production reduced by 50%	Leach et al. 1979
Cadmium	Mammals		350	toxicity threshold	Cooke and Johnson 1996
Cadmium	Rat		123	concentration associated with severe renal damage	Itokawa et al. 1974
Cadmium	White-tailed Deer	0.37±0.03		mean±SE for 190 individuals collected by hunters in 15 counties in IL. Maximum observation = 6.5 mg/kg dry wt. No effects noted.	Woolf et al. 1982
Cadmium	White-tailed Deer	1.71±0.28 (n=59)	16.5±5.6 (n=7)	mean±SE for individuals collected by hunters in throughout CT. Maximum liver and kidney = 9.1 and 48 mg/kg dry wt., respectively. No effects noted.	Musante et al. 1993
Cadmium	White-tailed Deer	0.8-2.6	20.9-39	Range of means from multiple regions in Quebec. Maximum liver and kidney = 10 and 74 mg/kg dry wt., respectively. No effects noted.	Crete et al. 1987
Cadmium	White-tailed Deer	11.6	93.9	Geometric mean of concentrations observed <8 km away from zinc smelter in PA.	Sileo and Beyer 1985
Cadmium	White-tailed Deer	4.2	55.9	Geometric mean of concentrations observed 10-20 km away from zinc smelter in PA.	Sileo and Beyer 1985
Cadmium	White-tailed Deer	1.9	15.5	Geometric mean of concentrations observed >100 km away from zinc smelter in PA.	Sileo and Beyer 1985
Copper	Sheep	30 - 150		normal	Buck 1978
Copper	Sheep	>450		toxicity threshold	Buck 1978
Copper	White-tailed Deer	109±5		mean±SE for 190 individuals collected by hunters in 15 counties in IL. Maximum observation = 456 mg/kg dry wt. No effects noted.	Woolf et al. 1982
Copper	White-tailed Deer	45.7±14.0		mean±SE for 19 individuals collected near uranium mine in TX. Maximum observation = 194.5 mg/kg dry wt. No effects noted.	King et al. 1984

Copper	White-tailed Deer	49.3±19.4		mean±SE for 17 individuals collected at control area in TX. Maximum observation = 257 mg/kg dry wt. No effects noted.	King et al. 1984
Copper	White-tailed Deer	190	18.2	Geometric mean of concentrations observed <8 km away from zinc smelter in PA.	Sileo and Beyer 1985
Copper	White-tailed Deer	82	20.6	Geometric mean of concentrations observed 10-20 km away from zinc smelter in PA.	Sileo and Beyer 1985
Copper	White-tailed Deer	106	18.2	Geometric mean of concentrations observed >100 km away from zinc smelter in PA.	Sileo and Beyer 1985
Copper	Chicken	184		concentration associated with NOAEL dose level	Mehring et al. 1960
Copper	Chicken	595		concentration associated with LOAEL dose level	Mehring et al. 1960
Lead	American Robin	10.5	25	collected from urban area - no effects noted	Getz et al. 1977
Lead	American Robin	2.4	7.3	collected from rural area - no effects noted	Getz et al. 1977
Lead	Red-winged Blackbird	52.6 [c]	88 [b]	concentration associated with 50% mortality	Beyer et al. 1988
Lead	Brown-headed Cowbird	80.6 [c]	240 [b]	concentration associated with 50% mortality	Beyer et al. 1988
Lead	Common Grackle	131.8 [c]	560 [b]	concentration associated with 50% mortality	Beyer et al. 1988
Lead	Mallard	120 [c]	272 [b]	concentration associated with 50% mortality	Beyer et al. 1988
Lead	Northern Bobwhite	340 [c]	740 [b]	concentration associated with 50% mortality	Beyer et al. 1988
Lead	Eastern Screech-owl	68.75 [c]	132 [b]	concentration associated with 50% mortality	Beyer et al. 1988
Lead	Yellow-billed Cuckoo	16.7 - 75.8 [a]	56 - 84 [b]	concentrations from three birds collected near zinc smelter	Beyer et al. 1985
Lead	Mammals	5	10	no toxicity noted below these levels	Ma 1996
Nickel	mice	2.3 [a]	4 [b]	concentration after lifetime exposure to 5 ppm in water - no effects observed.	Schroeder et al. 1964
Nickel	Rat	0.23 [a]	0.56 [b]	concentration after 2 yrs. on control diet - no effects observed	Ambrose et al. 1976
Nickel	Rat	2.0 [a]	16.6 [b]	concentration after 2 yrs on 2500 ppm diet. LOAEL for reproductive effects = 1000 ppm	Ambrose et al. 1976
Nickel	White-tailed Deer	3.6±0.7		mean±SE for 190 individuals collected by hunters in 15 counties in IL. Maximum observation = 97 mg/kg dry wt. No effects noted.	Woolf et al. 1982
Zinc	Peach-faced lovebirds	75 - 156		died from zinc poisoning	Reece et al. 1986

Zinc	Peach-faced lovebirds	12 - 33		unexposed to zinc; died from trauma	Reece et al. 1986
Zinc	Mallard	163.6 [a]	108 [b]	untreated birds	Gasaway and Buss 1972
Zinc	Mallard	1215 [a]	1652 [b]	birds consuming 3000 mg/kg diet for 60 days	Gasaway and Buss 1972
Zinc	Dog	51.5 - 97 [a]	36 - 92 [b]	normal	Eisler 1993
Zinc	Dog	1118 [a]	1180 [b]	zinc-poisoned	Eisler 1993
Zinc	Cattle	30 - 120		normal (>120 is considered elevated)	Eisler 1993
Zinc	Cattle	>500		lethal	Eisler 1993
Zinc	Sheep	144 - 165	84 - 150	normal	Eisler 1993
Zinc	Sheep	>250	>180	elevated	Eisler 1993
Zinc	Sheep	463 - 650	274 - 760	zinc-poisoned	Eisler 1993
Zinc	White-tailed Deer	70±2		mean±SE for 190 individuals collected by hunters in 15 counties in IL. Maximum observation = 252 mg/kg dry wt. No effects noted.	Woolf et al. 1982
Zinc	White-tailed Deer	256	310	Geometric mean of concentrations observed <8 km away from zinc smelter in PA. Maximum renal concentration (600 mg/kg dry wt.) from deer 4 km away from smelter. This deer had joint lesions similar to those seen in Zn-poisoned horses in the area.	Sileo and Beyer 1985
Zinc	White-tailed Deer	167	274	Geometric mean of concentrations observed 10-20 km away from zinc smelter in PA.	Sileo and Beyer 1985
Zinc	White-tailed Deer	132	145	Geometric mean of concentrations observed >100 km away from zinc smelter in PA.	Sileo and Beyer 1985

[a] wet weight reported; converted to dry wt using hepatic water content for mallards from Scanlon 1982

[b] wet weight reported; converted to dry wt using 75% renal water content from Beyer et al. 1988

[c] wet weight reported; converted to dry wt using hepatic water content from Beyer et al. 1988ranged from 75-156 mg/kg dry weight. Livers of

mg/kg dose level survived to 60 days. Liver and kidney zinc concentrations in the 3000 mg/kg dose level and the control birds are presented in Table 4.31.

Normal and toxic concentrations of zinc in the liver and kidneys of dogs, cattle and sheep were summarized by Eisler (1993). These values are presented in Table 4.31.

4.2.2.2 Biological surveys

While field studies of the effects of application of sewage sludge on wildlife have been performed, data are not available for all ecosystems or trophic groups considered in this assessment. The best represented ecosystems are the Douglas-fir and eastern deciduous forests. No wildlife field studies have as yet been performed in the loblolly pine plantation or rangeland ecosystems. Among endpoints, while effects of sewage sludge application on herbivorous and insectivorous small mammals (rodents and shrews), ungulates (deer and elk), and songbirds have been studied, effects on carnivores have not been considered. Summaries of available wildlife field studies are presented below.

Biological surveys in the Douglas-fir forest

The effects of sewage sludge application on wildlife within the Douglas-fir forest have been studied most extensively at the Charles Lathrop Pack Experimental Forest, southwest of Eatonville, Washington. Wildlife responses within various watersheds of the Pack forest have been documented following application of sludge from metropolitan Seattle.

Hegstrom (1986) and Hegstrom and West (1989) report the accumulation of cadmium, copper, lead, and zinc in liver and kidney of deer mouse (*Peromyscus maniculatus*), Trowbridge shrew (*Sorex trowbridgii*), and the shrew-mole (*Neurotrichus gibbsii*) following application of 51 Mg sludge/ha. All three species accumulated metals, with the highest levels observed in shrews and shrew-moles, particularly for cadmium. Cadmium and lead were elevated in all species from sludge treated sites. A summary of the organ metal concentrations in Trowbridge shrews reported by Hegstrom and West (1989) is presented in Table 4.32. Metal concentrations in soils at the Pack Forest estimated based on the 51 Mg/ha application rate and concentrations in sludge reported by Hegstrom and West (1989) are reported in Table 4.33.

To evaluate if cadmium concentrations resulted in hepatic or renal lesions, Hegstrom (1986) collected and analyzed additional Trowbridge shrews from sludge-treated and control stands. Cadmium in kidneys of these additional Trowbridge shrews (126 mg/kg dry wt.) was roughly equivalent to that which caused significant damage in rats (123 mg/kg dry wt.; Itokawa et al. 1974). Despite high levels of cadmium, no lesions were observed.

Evaluation of population characteristics for Trowbridge shrews found no conclusive indication of a population decline resulting from sludge application. However, the abundance of adult shrews collected from sludge-treated sites was approximately one-half of that seen in control sites (Hegstrom 1986). Lack of replication and changes in control sites limit the utility of these results.

Table 4.32. Summary of bioaccumulation data for wildlife endpoint species from sludge application field studies.

Species	Ecosystem	Analyte	Liver (mg/kg) dry	Kidney (mg/kg) dry	Treatment	Authors/Year
Trowbridge shrew	Douglas- fir forest	Cadmium	22	33	trt	Hegstrom and West, 1989
Trowbridge shrew	Douglas- fir forest	Cadmium	62	126	trt[a]	Hegstrom and West, 1989
Trowbridge shrew	Douglas- fir forest	Copper	36	46	trt	Hegstrom and West, 1989
Trowbridge shrew	Douglas- fir forest	Lead	0.5	4.3	trt	Hegstrom and West, 1989
Trowbridge shrew	Douglas- fir forest	Zinc	89	98	trt	Hegstrom and West, 1989
Trowbridge shrew	Douglas- fir forest	Cadmium	2.6	9	control	Hegstrom and West, 1989
Trowbridge shrew	Douglas- fir forest	Cadmium	3.9	13	control[a]	Hegstrom and West, 1989
Trowbridge shrew	Douglas- fir forest	Copper	26	37	control	Hegstrom and West, 1989
Trowbridge shrew	Douglas- fir forest	Lead	0.25	1.8	control	Hegstrom and West, 1989
Trowbridge shrew	Douglas- fir forest	Zinc	77	78	control	Hegstrom and West, 1989
black-tailed deer	Douglas- fir forest	Cadmium	0.5	3.3	trt	West et al. 1981
black-tailed deer	Douglas- fir forest	Copper	21	16	trt	West et al. 1981
black-tailed deer	Douglas- fir forest	Lead	0.2	0.8	trt	West et al. 1981
black-tailed deer	Douglas- fir forest	Zinc	97	96	trt	West et al. 1981
black-tailed deer	Douglas- fir forest	Cadmium	0.5	5.4	control	West et al. 1981
black-tailed deer	Douglas- fir forest	Copper	45	14	control	West et al. 1981
black-tailed deer	Douglas- fir forest	Lead	0.3	1.8	control	West et al. 1981
black-tailed deer	Douglas- fir forest	Zinc	91	93	control	West et al. 1981
American robin	Douglas- fir forest	Cadmium	5.3	18	Treated 1-2 yrs. post application	Milner, 1986
American robin	Douglas- fir forest	Copper	27	27	Treated 1-2 yrs. post application	Milner, 1986.
American robin	Douglas- fir forest	Lead	1.9	8.5	Treated 1-2 yrs. post application	Milner, 1986
American robin	Douglas- fir forest	Zinc	103	114	Treated 1-2 yrs. post application	Milner, 1986
American robin	Douglas- fir forest	Cadmium	0.77	2.8	Treated 5-10 yrs. post application	Milner, 1986
American robin	Douglas- fir forest	Copper	11	22	Treated 5-10 yrs. post application	Milner, 1986
American robin	Douglas- fir forest	Lead	1.9	2.7	Treated 5-10 yrs. post application	Milner, 1986
American robin	Douglas- fir forest	Zinc	79	87	Treated 5-10 yrs. post application	Milner, 1986
American robin	Douglas- fir forest	Cadmium	0.88	3.2	control	Milner, 1986
American robin	Douglas- fir forest	Copper	13	22	control	Milner, 1986
American robin	Douglas- fir forest	Lead	0.09	1	control	Milner, 1986
American robin	Douglas- fir forest	Zinc	79	91	control	Milner, 1986

Table 4.32. Summary of bioaccumulation data for wildlife endpoint species from sludge application field studies.

Species	Ecosystem	Analyte	Liver (mg/kg) dry	Kidney (mg/kg) dry	Treatment	Authors/Year
white-tailed deer	East. Decid. Forest	Cadmium	3.13	31.36	trt	Woodyard and Haufler, 1991
white-tailed deer	East. Decid. Forest	Copper	473	21.16	trt	Woodyard and Haufler, 1991
white-tailed deer	East. Decid. Forest	Nickel	1.53	0.99	trt	Woodyard and Haufler, 1991
white-tailed deer	East. Decid. Forest	Zinc	688	585	trt	Woodyard and Haufler, 1991

[a] Data from additional shrews collected to evaluate liver and kidney lesions.

Table 4.33. Estimated concentrations of metals in soils at sludge-treated sites from the literature

	Metal Concentrations in sewage sludge applied to forests			Estimated metal concentrations in Douglas-fir forest soils following application of sewage sludge		Estimated metal concentrations in Eastern deciduous forest soils following application of sewage sludge	
Chemical	**95% UCL from NSSS**	**Douglas-Fir forest[a]**	**Eastern deciduous forest[b]**	**This assessment[c]**	**Hegstrom and West (1989)[d]**	**This assessment[e]**	**Brockway (1988)[f]**
Cadmium	28.46	50	43.8	2.54	1.69	1.09	0.74
Copper	774.92	900	667.8	49.82	40.81	26.26	14.36
Lead	205.8	1200		82.09	49.99	17.02	
Nickel	84.6		37.4			13.11	11.66
Zinc	2602	2000	1085.6	181.82	183.16	90.26	45.35

[a] Data from Hegstrom and West (1989)
[b] Data from Brockway (1988)
[c] Calculated using assuming a sludge concentration equivalent to the 95% UCL from the NSSS, a 40Mg/ha application rate, mixing depth of 15 cm, bulk density of 1.1, and background soil concentraions reported in Table 4.1.2.
[d] Calculated using assuming sludge concentrations and a 51Mg/ha application rate from Hegstrom and West (1989), mixing depth of 15 cm, bulk density of 1.1, and background soil concentrations reported in Table 4.1.2.
[e] Calculated using assuming a sludge concentration equivalent to the 95% UCL from the NSSS, a 40Mg/ha application rate, mixing depth of 15 cm, bulk density of 1.4, and background soil concentraions reported in Table 4.1.4.
[f] Calculated using assuming sludge concentrations and a 9Mg/ha application rate from Brockway (1988), mixing depth of 15 cm, bulk density of 1.1, and background soil concentraions reported in Table 4.1.4.

Anderson (1983 and 1985) reports on the effects of sludge application on the food habits and reproductive success of Columbian black-tailed deer (*Odocoileus hemionus columbianus*) following application of 51 Mg sludge/ha. A shift in diet was observed, with deer consuming more grass in all seasons except summer and deer preferentially foraging in sludge-fertilized habitats. This behavior was attributed to an increase in the quantity and quality of cool-season grasses (Anderson 1983). Anderson (1985) also observed that deer in sludge-treated areas recruited twice as many fawns to the 6-month age class than did controls and that sludge application increased deer productivity by 40%. West et al. (1981) report approximately equivalent concentrations of cadmium, copper, lead, and zinc in liver and kidney of deer collected from sludge-treated and controls areas (values are presented in Table 4.32).

In a short-term (one field season), unreplicated (one treated and one untreated) study, Milner (1986) studied the response of birds to sludge-induced habitat alteration and heavy metal accumulation in birds following application of 51 Mg sludge/ha. Habitat responses to sludge application were not dramatic. While no major effects were observed, cover by wet-site and weedy plant species increased while that for dry-site plants decreased. While there were minimal differences between the sludge-treated and control sites in terms of use by birds in winter, abundance and richness of breeding birds was greater at the sludge-treated site.

To evaluate bioaccumulation by birds, Milner (1986) collected American robins, winter wrens (*Troglodytes troglodytes*), song sparrow (*Melospiza melodia*), and dark-eyed juncos (*Junco hyemalis*) from recent and 6-10 year old sludge-treated sites and as many as 6 control or reference areas. Concentrations of cadmium, copper, lead, and zinc in liver and kidney were analyzed in all birds. Insectivorous birds (robins and wrens) accumulated metals to a greater degree than did granivorous birds (sparrows and juncos). Maximum lead, cadmium, and zinc concentrations were seen in robins. For all four analytes, concentrations in liver and kidneys of robins from the sludge-treated area were greater than that within the control area. A summary of the organ metal concentrations measured in American robins reported by Milner (1986) is presented in Table 4.32.

In a recent study, Nickelson and West (1996) report renal cadmium concentrations in mice (*Peromyscus* sp.) and shrews (*Sorex* sp.) from sludge-treated forest lands in Washington. Sites had been treated 4, 11, or 15 years previously with varying amounts of sludge. Cadmium concentrations in soil ranged from 1.9 mg/kg at control sites to 17.5 mg/kg at a site treated with 500 Mg/ha sewage sludge containing 51 mg/kg cadmium 15 years prior. Renal cadmium concentrations were significantly greater in shrews than mice in all treatments. Shrews from sludge-treated sites had significantly higher renal cadmium concentrations than did shrews from control sites. Renal cadmium concentrations in shrews from the 11-year post application sites were approximately equivalent to those from shrews collected 2 years post-application by Hegstrom and West (1989). No adverse effects on shrew populations were observed.

Biological surveys in the eastern deciduous forest

Effects of sludge application were evaluated in five forest types in two counties in northern lower Michigan (Campa et al. 1986, Woodyard et al. 1986, Brockway 1988, Woodyard and Haufler 1991). Through a combination of field and laboratory studies, these authors considered bioaccumulation of cadmium, chromium, copper, nickel, and zinc by small mammals, deer, and woodcock. Metals in soils, selected wildlife forages, and small mammals were

monitored for three years in replicated sludge-treated and untreated plots. In addition, a laboratory study of a soil-to-macroinvertebrate-to-vertebrate insectivore food chain was evaluated.

In the field study, sewage sludge was applied to a 4-year-old jack pine clear-cut and stands of 10-year-old aspen, 70-year-old oak, 50-year-old jack and red pines, and 50-year-old mixed hardwoods. Sites received from 8 to 10 Mg sludge/ha. Metal concentrations in forest soils estimated based on a 9 Mg/ha application rate and concentrations in sludge reported by Brockway (1988) are reported in Table 4.33. Metal concentrations in vegetation and small mammals from each site were measured. Metal concentrations were also measured in 3 deer from the aspen stand in 1982 (values are presented in Table 4.32). The authors observed that while some plant species accumulated metals immediately after sewage sludge application, by the second growing season, concentrations had returned to background levels. Small mammals collected (all rodents) displayed few differences in metal concentrations between sewage sludge-treated and untreated areas. The authors state that small mammals do not appear to accumulate metals in sludge-treated sites, but note that insectivores such as shrews and moles were not collected, and results of this study may not be indicative of accumulations and risks that these species may experience.

In the laboratory food-chain portion of this study, earthworms were raised in sewage sludge-treated soils and fed to American woodcock (*Scolopax minor*) for 30 days. Woodcock fed sludge-exposed earthworms accumulated cadmium in liver and kidney to levels two times greater than controls. Chromium, copper, and nickel were not accumulated. Zinc was observed at higher concentrations only in kidneys among sludge-exposed birds.

In a related study, Campa et al. (1986) studied the use of sludge-treated aspen stands by white-tailed deer and elk *(Cervus elaphus)*. Sludge application increased the nutrient content of vegetation and as a result, both species browsed more heavily in sludge-treated sites than in untreated areas.

Biological surveys at mine reclamation sites

Sewage sludge has frequently been used to facilitate the reclamation of mined lands, and studies have been performed to evaluate potential bioaccumulation of contaminants. Dressler et al. (1986) measured the bioaccumulation of heavy metals (Cd, Cu, Co, Cr, Pb, Ni, and Zn) in cottontail rabbits (*Sylvilagus floridanus*) inhabiting a reclaimed strip mine in Pennsylvania that had received a one-time application of 134 Mg/ha sewage sludge. While concentrations of metals were elevated in vegetation relative to controls, with the exception of zinc in bones, metal concentrations in all tissues were not different from those observed in controls. At the same site, Alberici et al. (1989) found that heavy metal concentrations in tissues of meadow voles (*Microtus pennsylvanicus*) were equivalent to or lower than those at the control sites.

Gaffney and Ellertson (1979) report the concentrations of cadmium, lead, and copper in liver, kidney, brain, and muscle of adult and juvenile red-winged blackbirds from a reclaimed mine in Illinois that had been treated with 426 to 997 dry tons sludge/ha. Cadmium was elevated in kidneys of adults and zinc in livers of juveniles from the sewage sludge-treated area relative to controls.

Biological surveys at Miami University, Ohio

Since 1978, researchers at Miami University (OH) have studied the effects of repeated applications of processed sewage sludge (Milorganite) to an old field ecosystem. A series of papers have been published covering topics ranging from impacts of sludge on metal bioaccumulation (Anderson et al. 1982, Levine et al. 1989, Brueske and Barrett 1991), on insects (Larsen et al. 1994), on community structure and function (Maly and Barrett 1984), on population dynamics of meadow voles (Anderson and Barrett 1982), on behavior of meadow voles (Anderson 1982, Tilghman-Hall et al. 1990), and on efficacy of remedial actions (Benninger-Truax and Taylor 1993). In terms of effects on wildlife, studies have focused on behavioral responses and population dynamics of meadow voles and bioaccumulation in voles and the least shrew (*Cryptotis parva*; Brueske and Barrett 1991).

Anderson et al. (1982) found that following 2 years of applying 8.9 Mg/ha Milorganite, concentrations of cadmium in liver and kidneys of meadow voles were significantly higher than those observed in individuals from control sites. Mean cadmium concentrations from treated sites ranged from 0.1-1.1 mg/kg liver wet weight and 0.4-6.6 mg/kg kidney wet weight; concentrations from control sites were 0.01-0.04 mg/kg liver and 0.01-1.6 mg/kg kidney. Liver and kidney copper was elevated in some age/sex groups from treated sites. No treatment differences for lead or zinc were observed. While voles were found to accumulate metals in sludge-treated sites, no adverse effects on population parameters were observed (Anderson and Barrett 1982).

Levine et al. (1989) report that following 10 years (1978 through 1987) of annual sludge application, while cadmium and copper concentrations in soil increased over time, cadmium concentrations in liver and kidneys of voles at sludge-treated sites did not increase, and copper concentrations in the organs decreased. While concentrations of copper in liver and cadmium and copper in kidneys of voles from sludge-treated sites did not differ from those observed among control-site voles, liver cadmium was elevated among sludge-treated voles.

In behavioral studies of voles from sludge-treated sites, while Anderson (1982) observed normal behavior patterns, Tilghman-Hall et al. (1990) observed hyperactivity among female voles from sludge-treated areas. No effects were observed among males. Because concentrations of cadmium were significantly higher in brains and livers of voles from sludge-treated areas relative to controls, the authors attributed the behavioral effects to sex-related responses to cadmium toxicity.

Biological surveys at sites treated with paper mill sludge

While chlorinated dioxins and dibenzofurans are detected occasionally in municipal sewage sludge, their occurrences and concentrations are very low (maximum concentration of TCDD from NSSS: 0.00086 mg/kg; Table 4.1), and they are not generally measured in sewage sludge field studies. In contrast, dioxins may be of significant concern in paper mill sludges, and field studies have been performed to evaluate potential impacts.

Field and laboratory studies were conducted to evaluate impacts of application of pulp/paper mill sludge on wildlife inhabiting red pine plantations in Wisconsin (ERT 1987). One year post sludge application, TCDD and TCDF concentrations were determined in soil, undepurated earthworms, and deer mouse (whole body). Earthworm, litter invertebrate, deer mouse, and songbird population surveys were conducted. Pathology and enzyme induction were

evaluated in American robins. Soil samples indicated that the soil/litter/sludge matrix contained 10.8 ppt TCDD (mean) and 106 ppt TCDF (mean). Earthworms averaged 35.8 ppt TCDD (TCDF was not measured). Deer mice averaged 15.3 ppt TCDD and 7.3 ppt TCDF (whole body). No chemically-induced pathologies were observed among field collected birds and mammals. No differences were observed among reproductive or population parameters among birds or mammals at treated and control sites. Earthworm, deermouse, and insectivorous bird populations were generally higher in sludge-treated sites. Robins from sludge-treated areas had elevated liver enzyme (AHH and EROD) levels, indicating exposure to dioxin or other pollutants, but histopathology, reproduction, and population data indicated no discernable effects. The conclusion of the study was that risks to wildlife from dioxin in paper mill sludge were very low.

Vera and Servello (1994) studied the effects of application of paper mill sludge on wildlife in spruce-fir stands in Maine. Forty-five dry metric tons of sludge containing 21.3 ppt TCDD and 7.9 ppt TCDF were applied to four forest stands. Breeding birds, small mammals, invertebrates, and habitat were sampled. Invertebrate abundance did not differ between treated and untreated sites. Small mammal abundance differed among years of the study, but was comparable at treated and untreated sites. Breeding bird density did not differ among years and was not affected by sludge treatment. The overall conclusion of the study was that sludge application had no adverse effect on the breeding bird or small mammal community.

4.2.2.3 Toxicity tests

Results of laboratory toxicity tests of field-collected media are available only for shrews. To evaluate uptake and transfer of heavy metals at sludge-treated sites to secondary consumers, Brueske and Barrett (1991) fed least shrews diets containing earthworms collected from sludge-treated and untreated sites for 14 d. (Sludge-treated sites had received 8.96 Mg/ha of Milorganite per year for 11 years.) While both treatment and control groups consumed equivalent amounts of food/d, shrews in the treatment group experienced a statistically significant loss of body weight (mean=27.6%). Cadmium, copper, lead, and zinc all accumulated to significantly higher levels in the livers of treated shrews, as compared to control shrews. No significant accumulations in kidneys were observed. A summary of the organ metal concentrations measured in least shrews reported by Brueske and Barrett (1991) is presented in Table 4.34.

4.2.3 Risk Characterization for Wildlife

In the Risk Characterization, results of the exposure assessment (Sect. 4.2.1) and effects assessment (Sect. 4.2.2) are integrated to estimate risks (the likelihood of effects given the exposure) based on each line of evidence, and then a weight of evidence inference logic is applied to determine the best estimate of risk to each assessment endpoint. In an ideal, comprehensive risk assessment
there are three lines of evidence: literature-derived single chemical toxicity data (which indicate the toxic effects of the concentrations measured in site media), biological surveys of the affected system (which indicate the actual state of the receiving environment), and toxicity tests with ambient media (which indicate the toxic effects of the concentrations measured in site media). A summary of the lines of evidence available to assess risks from sludge application to wildlife endpoints in each ecosystem is presented in Table 4.35. Single chemical toxicity data were available for all wildlife endpoints and ecosystems. Biological survey data were available for

Table 4.34. Summary of bioaccumulation data for Least Shrew from Brueske and Barrett (1991). Shrews consumed diets containing earthworms from sludge-treated and untreated sites for 14 d. Body weight of shrews consuming sludge-treated diet decreased by 27.6%			
Analyte	**Concentration in Liver (mg/kg dry)**	**Concentration in Kidney (mg/kg dry)**	**Diet**
Cadmium	0.44	0.39	Sludge-treated
Cadmium	0.18	0.32	control
Copper	5.22	5.97	Sludge-treated
Copper	3.59	5.4	control
Lead	0.34	0.62	Sludge-treated
Lead	0.21	0.54	control
Zinc	25.31	20.44	Sludge-treated
Zinc	15.65	17.09	control

Table 4.35. Summary of lines of evidence available to assess risks of sludge application to wildlife endpoints

		Lines of Evidence Available		
Ecosystem	**Endpoint Species**	**Single Chemical Toxicity Data**	**Biological Survey Data**	**Toxicity Test Data**
Douglas-fir Forest	Trowbridge Shrew	available	available	available
	Black-tailed Deer	available	available	
	Red Fox	available		
	American Robin	available	available	
Loblolly Pine Plantation	Short-tailed Shrew	available		available
	White-tailed Deer	available		
	Red Fox	available		
	American Robin	available		
Eastern Deciduous Forest	Short-tailed Shrew	available		available
	White-tailed Deer	available	available	
	Red Fox	available		
	American Robin	available		
Semiarid Rangeland	Desert Shrew	available		available
	Mule Deer	available		
	Kit Fox	available		
	Western Meadowlark	available		

shrews, deer, and robins in the Douglas-fir forest and for white-tailed deer in eastern deciduous forests. Toxicity test data were available for shrews in all ecosystems.

Procedurally, the risk characterization is performed for each assessment endpoint by 1) screening all measured contaminants against toxicological benchmarks and background concentrations (if available), 2) estimating the effects of the contaminants retained by the screening analysis, 3) estimating the toxicity of the ambient media based on the media toxicity test results, 4) estimating the effects of exposure on the endpoint biota based on the results of the biological survey data, 5) logically integrating the lines of evidence to characterize risks to the endpoint, and 6) listing and discussing the uncertainties in the assessment. A detailed discussion of methods and the approach to risk characterization is presented in Suter et al. (1995).

4.2.3.1 Single chemical toxicity data

Screening exposure estimates

To determine if the contaminant exposures experienced by wildlife endpoints in each ecosystem are potentially hazardous and to identify COPECs, the screening exposure estimates for each exposure scenario (e.g. single 40 Mg/ha application of sludge containing the 95% UCL on the mean NSSS concentration, and background soil concentration) were compared to estimated NOAELs and LOAELs from Table 4.30. To quantify the magnitude of hazard, a hazard quotient (HQ) was calculated where: HQ = exposure/NOAEL or LOAEL. Hazard quotients greater than 1 indicate that individuals may be experiencing exposures that are in excess of NOAELs or LOAELs. While exceeding the NOAEL suggests that adverse effects are possible, exceeding the LOAEL suggests that adverse effects are likely. Hazard quotients and screening exposure estimates for wildlife endpoints within the Douglas-fir ecosystem are presented in Tables 4.12 through 4.15, endpoints within the loblolly pine plantation ecosystem are presented in Tables 4.16 through 4.19, endpoints within the eastern deciduous forest are presented in Tables 4.20 through 4.23, and endpoints within the semiarid rangeland ecosystem are presented in Tables 4.24 through 4.27. A summary of the comparison of the exposure estimates to LOAELs is presented in Table 4.36.

No risks from exposure to arsenic, cadmium, copper, lead, nickel, selenium, TCDD, PCB-1248, PCB-1254, and PCB-1260 were identified for any wildlife endpoint in any ecosystem. Among endpoints, no risks from any contaminant were identified for fox in any ecosystem under any exposure scenario (Table 4.36). Potential risks to deer were identified from molybdenum under the NSSS and background exposure scenarios in the Douglas-fir forest; risks from other contaminants were not identified for deer in any other ecosystem or exposure scenario (Table 4.36).

Under the NSSS and background exposure scenarios, mercury presents a risk to American robins in the Douglas-fir forest, loblolly pine, and deciduous forests, respectively (Table 4.36). Mercury presents potential risks to shrews in the Douglas-fir, deciduous forest, and rangeland ecosystems under the NSSS but not background exposure scenarios. Zinc presents a potential risks to robins in the Douglas-fir ecosystem under the NSSS scenario but not under the background scenario (Table 4.36).

Table 4.36. Summary of comparison of screening exposure estimates to LOAELs. Contaminants listed are those where HQs>1 were observed. Mercury assumed to be 100% methyl mercury.

		Exposure Scenario	
Ecosystem	**Endpoint Species**	**NSSS 95% UCL**	**Background Soil**
Douglas-fir Forest	Trowbridge Shrew	Hg	none
	Black-tailed Deer	Mo	Mo
	Red Fox	none	none
	American Robin	Hg, Zn	Hg
Loblolly Pine Plantation	Short-tailed Shrew	none	none
	White-tailed Deer	none	none
	Red Fox	none	none
	American Robin	Hg	Hg
Eastern Deciduous Forest	Short-tailed Shrew	Hg	none
	White-tailed Deer	none	none
	Red Fox	none	none
	American Robin	Hg	Hg
Semiarid Rangeland	Desert Shrew	Hg	none
	Mule Deer	none	none
	Kit Fox	none	none
	Western Meadowlark	none	none

Monte Carlo exposure estimates

To evaluate the influence of variability of parameters in the exposure model on exposure estimates, Monte Carlo simulations were performed for all endpoints and contaminants under the NSSS exposure scenario where HQs>1 were observed (Table 4.29). By superimposing NOAEL and LOAEL values on the exposure distributions generated from the Monte Carlo simulation, the likelihood of an individual experiencing potentially hazardous exposures can be estimated, and the magnitude of risk may be determined. Interpretation of the comparison of exposure distributions to NOAELs and LOAELs is described below:

Comparison	Meaning	Risk-based Interpretation
NOAEL>80th percentile of exposure distribution	less than 20% of exposures> NOAEL	Adverse effects to individuals are highly unlikely
NOAEL<80th percentile<LOAEL	More than 20% of exposures > NOAEL, but less than 20% of exposures> LOAEL	Individuals experiencing exposures at the high end of the distribution may experience adverse effects
LOAEL<80th percentile of exposure distribution	More than 20% of exposures> LOAEL	Adverse effects on individuals are likely

NOAEL and LOAEL values were overlain on inverse cumulative probability plots of each exposure distribution for each species and contaminant for which Monte Carlo simulations were performed (Figs 4.5 to 4.8). The entire exposure distribution of black-tailed deer to molybdenum in the Douglas-fir ecosystem fell below the LOAEL and less than 5% of the distribution exceeded the NOAEL (Fig. 4.5). This observation suggests that molybdenum exposure is unlikely to present a risk to black-tailed deer in the Douglas-fir forest. The LOAEL for zinc to American robins was less than the 20th percentile of the exposure distribution in the Douglas-fir forest, while the NOAEL fell near the minimum of the distribution (Fig. 4.6). As a consequence, zinc exposure presents a potential risk to American robins in this ecosystem

The likelihood of mercury exposure experienced by American robins exceeding the LOAEL for methyl mercury ranges from 60% for birds in the Douglas-fir forest to about 88% for birds in the deciduous forest or loblolly pine plantation (Fig. 4.7). However, because the >99.9% of mercury in sewage sludge is expected to be inorganic (Gilmour and Bloom 1994), comparison of exposure estimates to toxicity values for methyl mercury are likely to grossly overestimate risks from mercury. Comparison to the NOAEL and LOAEL for mercuric chloride indicates that the maximum exposure expected in any ecosystem was less than the NOAEL (Fig. 4.7), therefore mercury is unlikely to present a significant risk to American robins in any ecosystem.

Among shrews, while likelihood of mercury exposure exceeding the lowest methyl mercury NOAEL ranged from less than 5% for desert shrews in the semiarid range ecosystem to about 75% for Trowbridge shrews in the Douglas-fir ecosystem, the maximum exposure for shrews in any ecosystem did not exceed the lowest methyl mercury LOAEL and was at least an order of magnitude below the lowest mercuric chloride NOAEL (Fig. 4.8). The conclusion, therefore is that mercury does not present a risk to shrews in any ecosystem.

Effects of retained contaminants

Zinc. The avian NOAELs and LOAELs for zinc were derived from a study of white leghorn hens fed 20, 200, or 2000 ppm supplemental zinc (as zinc sulfate) in their diet for 44 weeks (Stahl et al. 1990). While no adverse effects were observed among hens consuming 20 and 200 ppm zinc, egg hatchability was <20% of controls among hens consuming 2000 ppm supplemental zinc. Because the study was greater than 10 weeks in duration and considered exposure during

reproduction, the 200 ppm dose was considered a chronic NOAEL and the 2000 ppm dose was considered a chronic LOAEL. Based on the results of Stahl et al. (1990), individual American robins experiencing exposure at or above the LOAEL are likely to reduced hatching success of eggs.

4.2.3.2 Toxicity tests

Toxicity test data are only available for shrews. Brueske and Barrett (1991) collected earthworms from old-field study sites in Ohio that had been treated annually with 8.96 Mg/ha sewage sludge in the form of Milorganite for 11 years. Earthworms were formulated into a diet and fed to least shrews for 14 days. Accumulation of cadmium, copper, lead, and zinc in liver and kidneys were measured. A summary of the results of this study are presented in Section 4.2.2.3.

With only one exception, concentrations of cadmium, copper, lead, and zinc in the livers and kidneys of treatment shrews from the Brueske and Barrett (1991) study were lower than concentrations observed among Trowbridge shrews collected from both control and sludge-treated sites in the Douglas-fir ecosystem (Table 4.37). The sole exception was lead in the liver of Trowbridge shrews from control sites.

Table 4.37. Comparison of bioaccumulation data for least shrew (Brueske and Barrett 1991) to organ metal values for Trowbridge shrews (Hegstrom and West 1989).

		Least Shrew		Trowbridge Shrew	
Analyte	**Diet/Sample Location**	**Liver (mg/kg dry)**	**Kidney (mg/kg dry)**	**Liver (mg/kg dry)**	**Kidney (mg/kg dry)**
Cadmium	Treatment	0	0	22	33
	Control	0	0	3	9
Copper	Treatment	5	6	36	46
	Control	4	5	26	37
Lead	Treatment	0	1	1	4
	Control	0	1	0	2
Zinc	Treatment	25	20	89	98
	Control	16	17	77	78

Differences in organ metal concentrations between least and Trowbridge shrews are most likely due to length of exposure. While least shrews were exposed to elevated metals for two weeks, the organ concentrations observed in Trowbridge shrews reflect life-long exposures. For this reason, the significance of the higher organ metal concentrations of Trowbridge shrews relative to least shrews cannot be determined.

Daily heavy metal exposure estimates for least shrews in the Brueske and Barrett (1991) study were calculated for comparison to estimated exposure for shrews in other ecosystems (Table 4.38). Body weight values were obtained from Silva and Downing (1995; 0.0047 kg). Food ingestion rates for least shrews were reported in Brueske and Barrett (1991). LOAELs for least shrews were calculated using the methods outlined in Section 4.2.2.1. While estimated

exposures were all lower than literature-derived toxicity values based on reproductive effects (Table 4.38), Brueske and Barrett (1991) observed that despite consuming the equivalent amount of food, shrews on the treatment diet lost >27% of body weight during the course of the study. It was suggested that the weight loss was a result of lead toxicity (Brueske and Barrett 1991).

Table 4.38. Summary of estimated heavy metal exposures experienced by least shrews in Brueske and Barrett (1991)

Analyte	Conc. in Contaminated Diet[a] (mg/kg)	Body Weight[b] (kg)	Food Ingestion Rate[a] (kg/d)	Exposure (mg/kg/d)	LOAEL	LOAEL HQ
Cd	28.87	0.0047	0.00248	15.234	28.336	0.54
Cu	10.13	0.0047	0.00248	5.345	58.8	0.09
Pb	10.56	0.0047	0.00248	5.572	235.01	0.02
Zn	183.26	0.0047	0.00248	96.699	940.0	0.10
Analyte	**Concentration in Control Diet[a] (mg/kg)**					
Cd	8.22	0.0047	0.00248	4.337	28.336	0.15
Cu	7.59	0.0047	0.00248	4.005	58.8	0.07
Pb	6.15	0.0047	0.00248	3.245	235.01	0.01
Zn	131.18	0.0047	0.00248	69.218	940.0	0.07

[a] Values from Brueske and Barrett 1991.
[b] Value from Silva and Downing 1995.

Exposure estimates for least shrews were also compared to estimated exposures for shrews in other ecosystems (Table 4.39). Estimated exposure of short-tailed shrews in the loblolly plantation ecosystem to all four metals (Cd, Cu, Pb, and Zn) was lower than the highest experienced by least shrews. In contrast, exposure of shrews from the other three ecosystems (Douglas-fir, eastern deciduous forest, and rangeland) to copper, Pb, and zinc exceeded that experienced by least shrews (Table 4.39), while exposure to cadmium was lower than that for least shrews. If it is assumed that cadmium contributes negligibly to the effects observed by Brueske and Barrett (1991), these results suggest that shrews in the Douglas-fir, eastern deciduous forest, and rangeland may experience adverse effects comparable to those experienced by least shrews (e.g. significant weight loss).

4.2.3.3 Biological surveys

Douglas-fir forest

Field surveys of impacts of sludge application on Trowbridge shrews, black-tailed deer, and American robins were performed in the Pack Forest in the Douglas-fir ecosystem. Data from Hegstrom (1986), Hegstrom and West (1989), Nickelson and West (1996) indicate that while Trowbridge shrews from sludge treated areas accumulate metals to higher levels than observed in control sites, no adverse effects were apparent. Despite maximum renal cadmium concentrations (Table 4.32) roughly equivalent to that which caused significant damage in rats (Table 4.31), no

lesions were observed. Copper, lead, and zinc concentrations in liver and kidneys of shrews from sludge-treated areas (Table 4.32) were also lower than concentrations associated with toxicity (Table 4.31). Population analysis indicated that abundance of adult shrews collected form sludge treated sites was approximately 1/2 of that seen in control sites (Hegstrom 1986). Lack of replication, changes in control sites, and high variability of small mammal populations limit the utility of these results and does not allow strong conclusions concerning effects from sludge application to be made.

Table 4.39. Comparison of exposure estimates for shrews in assessment ecosystems to that experienced by least shrews in Brueske and Barrett (1991).

	Estimated exposures for Least Shrews (mg/kg/d)		Estimated exposures for shrews in assessment ecosystems (mg/kg/d)			
Analyte	**Control**	**Treatment**	**Douglas-fir: Trowbridge shrew**	**Loblolly Pine: Short-tailed shrew**	**Eastern Deciduous Forest: Short-tailed shrew**	**Semiarid Rangeland: Desert shrew**
Cadmium	4.337	15.234	4.041	6.159	7.959	1.395
Copper	4.005	5.345	88.504	5.025	5.94	22.624
Lead	3.245	5.572	130.029	3.336	7.529	22.456
Zinc	69.218	96.699	2547.6	75.612	142.237	643.855

Anderson (1983 and 1985) report that black-tailed deer responded positively to sludge application: deer preferentially foraged in sludge-treated areas, and reproductive success increased. Concentrations of metals (Cd, Cu, Pb, and Zn) in liver and kidney of deer from sludge treated areas were equivalent to those from control areas (West et al. 1981; Table 4.32) and below levels concentrations associated with toxicity (Table 4.31).

Among birds, Milner (1986) found no dramatic responses to sludge application. While minimal differences were observed in the use of sludge-treated sites in winter, abundance and richness of breeding birds were greater in sludge-treated areas. Evaluation of metal bioaccumulation by birds indicated greater accumulation by insectivores as compared to granivores. The highest levels were observed in American robins. Concentrations of cadmium, copper, and lead in liver and kidney of robins (Table 4.32) were lower than concentrations associated with toxicity (Table 4.31). Concentrations of zinc in liver from robins from both treated and control areas were equivalent to levels associated with mortality among peach-faced lovebirds (Reece et al. 1986) but approximately 1/10 of concentrations associated with mortality in mallards (Table 4.31). Because both lovebirds and mallards are taxonomically distant from robins, concentrations that are toxic to these species may not be indicative of potential effects to robins.

Loblolly pine plantation

No field studies for wildlife were conducted in this ecosystem. Therefore this line of evidence is unavailable.

Eastern deciduous forest

Campa et al. (1986) report that white-tailed deer and elk foraged preferentially in sludge-treated areas in Michigan. Bioaccumulation of cadmium, copper, nickel, and zinc was evaluated in three white-tailed deer (Table 4.32) collected from an 10-year-old aspen stand that had been treated with 9.98 Mg/ha one year previously (Campa et al. 1986, Woodyard and Haufler 1991). Because samples were not collected from deer obtained from sites that had not received sludge, it cannot be determined if the metal levels in white-tailed deer are associated with sludge application or simply reflect background exposure. However, concentrations of cadmium, copper and zinc in liver and kidney of white-tailed deer from this study were markedly greater than those observed in black-tailed deer from both sludge-treated and control sites in the Douglas-fir ecosystem (Table 4.32).

When compared to other literature data, cadmium and nickel concentrations in organs of white-tailed deer from the sludge-treated site (Campa et al. 1986; Table 4.32) did not exceed concentrations associated with toxicity or those observed among white-tailed deer from other locations in the United States (Table 4.31). In contrast, copper concentrations in liver of deer from the sludge-treated site exceeded the maximum observed among deer in Illinois, were 1.84 to 2.43 times greater than the maximum observed among deer collected near a mining site in Texas, and exceeded concentrations associated with toxicity in sheep (Table 4.31). For zinc, concentrations in liver and kidney exceeded toxic concentrations for both cattle and sheep, concentrations in liver were over twice the maximum observed among deer in Illinois, and concentrations in kidneys were comparable to that observed in deer exhibiting symptoms of zinc-poisoning from near a smelting complex in Pennsylvania (Table 4.31).

Semiarid rangeland

No field studies for wildlife were conducted in this ecosystem. Therefore this line of evidence is unavailable.

Other field studies

While other field studies of the effects of sludge application on wildlife have been conducted and are summarized in Section 4.2.2.2, most are not applicable to the endpoints evaluated in this assessment. The exceptions are the studies of the impacts of forest applications of papermill sludges. These studies provide the only indications of potential effects due to contamination from chlorinated dioxins and dibenzofurans.

Field studies on the effects of application of papermill effluent on wildlife have been conducted in central Wisconsin (ERT 1987) and in Maine (Vera and Servello 1994). In the Wisconsin study, TCDD and TCDF were detected in soil, earthworms and small mammals (ERT 1987). American robins from treated sites displayed elevated liver enzyme activities. Despite the evidence of exposure to TCDD and TCDF, chemically induced pathologies were not observed, and reproductive and population parameters for both birds and small mammals were not affected. This study concluded that risks from exposure to contaminants in mill sludge were low. A similar lack of impact on birds and small mammals following application of papermill sludge was observed by Vera and Servello (1994).

4.2.3.4 Weight of evidence

The weight-of-evidence for the risks to wildlife associated with application of sewage sludge at 40 Mg/ha in the four ecosystems is outlined below.

Shrews

All three lines of evidence (e.g., single chemical toxicity data, field surveys, and media toxicity tests) were available for Trowbridge shews in the Douglas-fir forest; however only single chemical toxicity data and media toxicity test data were available for shrews in the other three ecosystems. Screening of exposure estimates against single chemical toxicity data suggests that while no contaminants present a risk to short-tailed shrews in the Loblolly plantation ecosystem, only one contaminant, mercury (assumed to be in the form of methyl mercury), may present a risk to shrews in the remaining three ecosytems. Because most mercury in sewage sludge is likely to be in the inorganic forms, comparison of exposure estimates to methyl mercury toxicity data overestimates potential risk. Because the maximum Monte Carlo estimates of exposure for shrews in the Douglas-Fir forest, eastern deciduous forest, and semiarid rangeland ecosystems were less than both the lowest methyl mercury LOAEL and lowest mercuric chloride NOAEL (Fig. 4.8), the conclusion from this line of evidence is that contaminants in municipal sewage sludge do not present a risk to shrews.

Field survey data for Trowbridge shrews indicated that metals accumulated to levels significantly greater than at control sites. However, despite maximum renal cadmium concentrations in shrews from sludge-treated sites being equivalent to levels associated with toxicity in rats, no renal lesions were observed. Additionally, copper, lead, and zinc concentrations in liver and kidneys of shrews from sludge-treated areas are all lower than concentrations associated with toxicity. Population analyses indicated that abundance of adult shrews was approximately 50% of that at control sites (Hegstrom 1986). However, due to limited replication, changes at control sites, and high variability of small mammal populations, a relationship of the observed population effect to sludge application is equivocal at best. The conclusion from this line of evidence is that sludge application is unlikely to present a risk to shrews.

One media toxicity test for shrews was available (Brueske and Barrett 1991). Concentrations of cadmium, copper, lead and zinc in liver and kidneys of Trowbridge shrews from both sludge-treated and control sites in the Douglas-fir forest exceeded those observed in least shrews in the toxicity test. Differences may be due to short exposure duration in the toxicity test (2 weeks). Field data for comparison to toxicity test results were not available from the other three ecosystems. While estimated exposures of shrews in the loblolly plantation ecosystem to copper, lead, and zinc, and cadmium were lower than the highest exposure experienced by least shrews in the toxicity test, exposures of shrews from the Douglas-fir, eastern deciduous forest, and rangeland ecosystems to copper, lead, and zinc exceeded those experienced by least shrews in the toxicity test. Because shrews in the toxicity test lost 27% of the body weight over the course of the test and contaminant exposures of shrews in the Douglas-fir, eastern deciduous forest, and rangeland ecosystems exceeded those among shrews in the toxicity test, weight loss due to contaminant exposure may occur among shrews at sludge-treated sites. Uncertainty about this conclusion is high however, due to the short duration of the toxicity test (it is not known if the weight loss would be sustained over an ecologically relevant time frame), because it is not known which chemical(s) and mechanisms are responsible for the observed effect, and because the comparability of observed effects among different species is not known. In addition, the ecological relevence of short-term weight loss on individual survival or

reproduction is not known. Therefore, the conclusion from this line of evidence is that sludge application is unlikely to present a risk to shrews.

Based on all three lines of evidence, the weight of evidence suggests that sludge application is unlikely to present a risk to shrews in any of the four ecosystems.

Deer

Although two lines of evidence (e.g., single chemical toxicity data, and field surveys) were available for deer in the Douglas-fir and eastern deciduous forests, only single chemical toxicity data were available for deer in the other two ecosystems. Screening of exposure estimates against single chemical toxicity data suggests that only one chemical (molybdenum) in one ecosystem (Douglas-fir forest) presents a risk to deer. However, because the maximum Monte Carlo estimate of molybdenum exposure for deer in the Douglas-Fir forest were less than the LOAEL (Fig. 4.5), the conclusion from this line of evidence is that contaminants in sewage sludge do not present a risk to deer in any ecosystem.

Field survey data showed that deer in the Douglas-fir forest did not significantly accumulate contaminants to levels higher than controls and that reproduction was enhanced at sludge-treated sites. In the eastern deciduous forests, deer preferentially browsed in sludge-treated forests. Because data from control sites were not collected, bioaccumulation of contaminants by deer in the eastern deciduous forest are difficult to interpret. Although concentrations of cadmium and nickel in organs are comparable to those for deer from sludge-treated Douglas-fir forests, those for copper and zinc exceed that observed in uncontaminated deer from other locations and are within the range of concentrations associated with adverse effects. The conclusion from this line of evidence is that sludge application is unlikely to present a risk to deer in the Douglas-fir forest but may present a risk to deer in the eastern deciduous forest.

The weight of evidence, based on two lines of evidence, suggests that sewage sludge application is unlikely to present a risk to deer in the Douglas-fir forest, loblolly pine plantation, or semiarid rangeland. Although sludge application may also present little risk to deer in the eastern deciduous forest, the problematic bioaccumulation data prevent a strong conclusion of no risk. Therefore it must be concluded that copper and zinc may present a risk to deer in the eastern deciduous forest.

Fox

Only one line of evidence, single chemical toxicity data, was available for foxes in all ecosystems. Because exposure estimates for all contaminants in sewage sludge did not exceed LOAELs, the weight of evidence suggests that sludge application is unlikely to present a risk to foxes in any of the four ecosystems.

Birds

While two lines of evidence (e.g., single chemical toxicity data, and field surveys) were available for American robins in the Douglas-fir forest, only single chemical toxicity data were available for birds in the other three ecosystems. Screening of exposure estimates against single chemical toxicity data suggests that zinc may present a risk to American robins in the Douglas-fir

forest and that mercury (assumed to be in the form of methyl mercury) may present a risk to American robins in the Douglas-fir forest, eastern deciduous forest and loblolly pine plantation ecosystems. Because most mercury in sewage sludge is likely to be in the inorganic forms, comparison of exposure estimates to methyl mercury toxicity data overestimates potential risk. Because the maximum Monte Carlo estimates of exposure for robins in the Douglas-Fir forest, eastern deciduous forest, and loblolly pine plantation ecosystems were less than the mercuric chloride NOAEL, the conclusion is that mercury in sewage sludge applied at 40 Mg/ha does not present a risk to birds. In contrast, because the LOAEL for zinc was less than the 20th percentile of the zinc exposure distribution for American robins in the Douglas-fir forest (Fig. 4.6), zinc may present a risk to birds in this ecosystem.

Field survey data showed no dramatic responses of birds to sludge application in the Douglas-fir forest, with the abundance and richness of breeding birds greater in sludge-treated areas. While concentrations of cadmium, copper, and lead in liver and kidney of robins were lower than concentrations associated with toxicity, zinc concentrations in liver from robins from both treated and control areas were equivalent to levels associated with mortality among peach-faced lovebirds but approximately 1/10 of concentrations associated with mortality in mallards. Because both lovebirds and mallards are taxonomically distant from robins, concentrations that are toxic to these species may not be indicative of potential effects to robins. In addition, liver zinc concentrations from both sludge-treated and control sites were equivalent. Consequently, the conclusion from this line of evidence is that sludge application does present a risk to birds.

The weight of evidence, based on a single line of evidence, suggests that sludge application is unlikely to present a risk to birds in the eastern deciduous forest, loblolly pine plantation, and semiarid rangeland. In the Douglas-fir forest, although single-chemical toxicity data suggest that zinc may present a risk to American robins, this conclusion is not supported by the field survey data. Because field survey data provide a stronger representation of effects, and have fewer uncertainties, than single chemical toxicity data, the weight of evidence for birds in the Douglas-fir forest is that risks from application of sewage sludge are unlikely.

Conclusions

The conclusions of the weight of evidence are that individual fox, shrews, American robins, and western meadow larks are not at risk from a one-time application of 40 Mg/ha of sewage sludge (containing contaminant concentrations reported in the NSSS) in any ecosystem, and individual deer are not at risk in the Douglas-fir forest, Loblolly pine plantation, or semiarid rangeland ecosystems. Limited bioaccumulation data suggests that white-tailed deer in the eastern deciduous forest may be at risk from copper and zinc. Because populations consist of individuals and individual-level risks are unlikely, population-level risks to all endpoints considered are also unlikely.

4.2.3.5 Relationship of effects of contaminants to effects of nutrients on wildlife

In addition to potentially toxic contaminants, sewage sludge contains plant nutrients which have been shown to increase plant growth and alter plant community composition and structure. These effects of application of sewage sludge on plants can also have indirect effects on the wildlife communities present at sludge-treated sites. The species composition and physical structure of plant communities are key determinants of habitat quality for many wildlife species, birds in particular. By altering the structure and composition of plant communities, sludge

application may make habitats more suitable for different species of wildlife, resulting in higher species richness or population abundance. For example, Milner (1986) observed that abundance and richness of breeding birds were greater in sludge-treated areas as compared to untreated areas. In another study, earthworm, deer mouse, and insectivorous bird populations were higher in paper-mill sludge-treated sites than at untreated sites (ERT 1987). Alternately, if decomposition of forest litter, which provides habitat and foraging locations for shrews, other small mammals, and many insectivorous birds, is accelerated with the addition of sewage sludge, as suggested in Sect. 5.4, these wildlife species could be adversely affected.

Changes in abundance and nutritional quality of plants at sewage sludge treated sites can also have direct effects on herbivorous wildlife. Greater quantity and higher quality of forage following sludge application can attract animals to use these sites and increase the growth and survival of these individuals. Such observations have been reported among deer at sludge-treated Douglas-Fir forest sites (Anderson 1983 and 1985) and Eastern Deciduous forest sites (Campa et al. 1986).

It is difficult to compare the potential effects of contaminants on wildlife to plant- and habitat-mediated effects associated with sludge application. While contaminant effects are likely to negative for all exposed species, plant and habitat mediated effects may be either positive or negative depending on the habitat requirements of each species. Many herbivorous wildlife species are likely to benefit from increased abundance and quality of food following sludge application. In contrast, species requiring open habitat with lower plant diversity are likely to be adversely affected by sludge application. Due to the minimal extent of contaminant risks estimated or observed at sludge-treated sites and because wildlife responses to plant and habitat-mediated effects of sludge application have been observed, it is likely that the plant and habitat-mediated effects of sludge application will predominate.

4.2.3.6 Uncertainties

Significant uncertainties are associated with the risk assessment for wildlife. These uncertainties are outlined below.

- **White-tailed deer bioaccumulation data.** Data concerning the bioaccumulation of contaminants by white-tailed deer at sewage sludge treated eastern deciduous forest sites consisted of three observations from a single location. No data from comparable untreated sites were collected. While these data are elevated relative to other reported data, the small sample size and lack of locally collected controls prevents strong conclusions.

- **Background concentrations.** Background concentrations of chemicals in soil were often responsible for a substantial fraction of the estimated toxicity. Background levels varied significantly among the ecosystems in this study. The spatial variability of background levels within an ecosystem and that within sludge-treated ecosystems in the United States are unknown.

- **Incorporation of sewage sludge.** The rate of incorporation of chemicals in sewage sludge into soils is unknown, as is the conservatism of the assumption that the sewage sludge is mixed in the surface 15 cm of soil.

- **Uptake models.** Generic uptake models were employed. These emodels did not incorporate aging of contaminants in soil or soil characteristics that may either increase or decrease uptake. As a consequence, contaminants in sludge may have been more or less available for uptake than assumed.

- **Body burden data.** Factors that affect the applicability of body burden toxicity data to wildlife endpoints in this assessment are unknown.

- **Shrews.** The applicability of results of the toxicity test on the least shrew to other shrews is unknown. In addition, ecological significance of short-term weight loss on survival and reproduction is unknown.

- **Bioavailability of contaminants.** The potential difference in bioavailability between chemicals in sewage sludge-treated soil and chemicals in untreated soil has been discussed previously. It is also notable that bioavailability may vary from sludge to sludge and from soil to soil. Media toxicity tests and soil-to-biota uptake factors rarely were derived from the particular ecosystems of concern.

- **Extrapolation of ambient media toxicity test data.** It has been assumed that media toxicity tests and biota surveys at application rates of 40 Mg/ha are comparable to the assumptions in this assessment. To determine if this is true, the concentrations of active chemicals (particular nutrients or contaminants) in sludge would have to be measured, as well as the ultimate concentrations in soil.

- **Extrapolation from published wildlife toxicity data**. Published toxicity studies were not available for all endpoint species considered. Consequently, inter-species extrapolation was necessary. Although it was assumed that toxicity could be estimated as a function of body size, the accuracy of this method is not known.

Additional extrapolation uncertainty exists for those contaminants for which data consisted of only LOAELs, or for which tests were subchronic in duration. In either case, an uncertainty factor of 10 was employed to estimate NOAELs or chronic data. The uncertainty factor of 10 may either over- or underestimate the actual LOAEL-NOAEL or subchronic-chronic toxicity relationship.

Toxicity of PCBs was evaluated using toxicity data from studies on Aroclor 1254. Because toxicity of PCB congeners can vary dramatically, the applicability of data for Aroclor 1254 to PCBs in general is unknown.

- **Single contaminant tests vs exposure to multiple contaminants in the field.** Although organisms in the four ecosystems are exposed to multiple contaminants concurrently, published toxicological values from which ORNL benchmarks were derived only consider effects experienced by exposures to single contaminants. Because some contaminants to which wildlife and plants are exposed can interact antagonistically, single contaminant studies may overestimate their toxic potential. Similarly, for those contaminants that interact additively or synergistically, single contaminant studies may underestimate their toxic potential.

- **Inorganic forms or species present in the environment.** Toxicity of metal species varies dramatically depending upon the valence state or form (organic or inorganic) of the metal. For example, Arsenic(III) and methyl mercury are more toxic to wildlife than arsenic(V) and inorganic mercury, respectively. The NSSS does not report species or form of contaminants in municipal sewage sludges. For wildlife, benchmarks used for comparison represented the more toxic species/forms of the metals (particularly for arsenic and mercury); thus potential toxicity in the ecosystems may be overestimated.

- **Contaminant concentrations in unanalyzed food types.** Uptake factors were not available for all food types consumed by the wildlife endpoint species. It was assumed that the uptake models for food types for which we had data were representative of those for which no data were available. Due to the different life histories and food types, contaminant burdens are likely to differ from the measured data. Therefore, assuming comparability among food types may either over or underestimate exposure.

4.2.3.7 Comparison of this risk assessment to that in Technical Support Document (TSD) (EPA 1992a)

While ecological risks associated with the land application of sewage sludge have been considered previously (TSD, EPA 1992a), direct comparisons of the results and conclusions are difficult due to differences in the methods employed, and the exposure scenarios, endpoints, and contaminants considered. The differences between these assessments are summarized in Table 4.40.

Table 4.40. Comparison of attributes of the ecological risk assessment for land application of sewage sludge reported in the TSD (1992) and those from this assessment

Attribute	TSD 1992	This Assessment
Ecosystems	Two generic ecosystems: - agricultural and - non-agricultural	Four specific ecosystems: - Douglas-fir forest, - Eastern Deciduous forest, - Loblolly Pine plantation, and - Semiarid Rangeland
endpoints	Two endpoints in each ecosystem: - Herbivore (represented by domestic animal or by deer mouse) - earthworm predator (represented by shrew)	Four endpoints in each ecosystem: - Herbivore (represented by deer), - Mammalian ground invertebrate feeder (represented by shrew) - Avian invertebrate feeder (represented by American robin or western meadowlark) - Mammalian predator (fox)

contaminants	- Up to nine contaminants considered for herbivores (As, Cd, Cr, Cu, Pb, Mo, Ni, Se, and Zn) - only three (Cd, Pb, and PCBs) considered for shrews.	Considered 11 contaminants (As, Cd, Cu, Pb, Hg, Mo, Ni, Se, Zn, TCDD, and PCBs) for all endpoints in each ecosystem.
Risk assessment methodology	- Risk estimate based on a single line of evidence (single chemical toxicity data)	- Risk estimate based on a multiple lines of evidence (e.g., single chemical toxicity data, field survey data, and media toxicity test data).
modeling approach	- Employed multiple modeling approaches. Most consisted of estimation of the application rate for contaminants (kg/ha) equivalent to maximal tolerable doses. Alternate approach based on estimation of application rate associated with toxic contaminant concentrations in organs. - Employed only deterministic models.	- Employed a single modeling approach to generate an estimate of oral exposure under a fixed application scenario which was then compared to NOAELs and LOAELs. - Employed both deterministic and probabilistic methods to incorporate variability and uncertainty.
background	- Background concentration subtracted from model and assumed not to contribute to risk.	- Background concentration included in models and assumed to contribute to risk.
Toxicity data employed	Toxicity data widely variable. Some values maximum tolerable doses, others levels associated with organ pathology. All data used interchangeably.	Used standardized toxicity data concerning NOAELs and LOAELs for survival and reproduction of exposed individuals.

4.3 RISKS TO PLANTS

4.3.1 Exposure Assessment for Plants

The general exposure assumptions for the ecological risk assessment are stated in Section 4.1. A critical assumption for plants is that constituents of sludge are evenly distributed through the top 15 cm of soil. The assumption is conservative if plant roots are exposed partly to uncontaminated soils beneath the sludge-amended zone and liberal for plant communities where feeder roots run through sludge that has not been incorporated into soil. The soil concentrations of contaminants to which the plants are assumed to be exposed are presented in Tables 4.2, Douglas-fir forest; 4.3, loblolly pine plantations; 4.4, deciduous forest; and 4.5, semi-arid rangeland. It should be noted that the uptake patterns, and therefore the exposure, of different plant species are expected to vary somewhat.

Deep-rooted trees and shallow-rooted grasses may be exposed to different ranges of contaminant concentrations, depending on the location of the source. In the absence of data on root distribution, in phytotoxicity and uptake studies it is assumed that the plants are rooted within the zone from which the soil was sampled for analysis. Foxx et al. (1984) note that the rooting depth of plants varies with species; nutrient and oxygen availability; soil water; soil temperature; presence of pathogens; and soil pore size, distribution and compaction. The addition of sewage sludge may impact rooting depth and density. For example, plant root growth was much lower on plots treated with 45 Mg/ha of sludge than on untreated plots in the semi-arid rangeland of New Mexico (Loftin and Aguilar 1994).

Bower (1984) suggests that the feeder roots in forests are found mostly in the top 30 cm, with root density and absorption decreasing exponentially from the surface. Foxx et al. (1984) reviewed literature on the rooting depths of native and crop plants that occur mostly west of the Mississippi River. The minimum, average, and maximum rooting depths for deciduous trees were 73, 332, and 3000 cm. For evergreens, the numbers were 10, 336, and 6096 cm. For perennial forbs, the depths were 2, 170, and 3932 cm. The shallowest-rooting group was the annual grasses, ranging from 5 to 110 cm with an average rooting depth of 52 cm. Roots for support were not distinguished from feeder roots, but it is likely that the assumption that plants are exposed exclusively to the sludge-treated soil in the top 15 cm is a conservative one.

In the ecosystems of concern in this assessment, rooting depth was rarely measured. In the rangeland, eighty-five percent of the root biomass of blue grama is in the top 20 cm of soil (Singh and Coleman 1977), but broom snakeweed has a much deeper rooting depth (McDaniel et al. 1982). On the other hand in the Everett gravelly silty loam at Pack Forest, Washington, many roots were observed as deep as 81 cm, but no distinction was made between roots that take up nutrients and structural roots (Greenleaf-Jenkins and Zasoski 1986).

When sludge is applied to the soil surface in forests or rangeland, the surfaces of foliage or stems of existing plants may be exposed to biosolids during the application process. Direct exposures of foliage to sewage sludge are not considered in this risk assessment.

4.3.2 Effects Assessment for the Plant Community

In an ideal ecological risk assessment, three lines of evidence are weighed: single chemical toxicity data, biological surveys of the affected system, and toxicity tests with ambient

media, in this case sewage sludge-treated soils. Single chemical toxicity data were available for most contaminants in sludge, usually from experiments with metal salts. Vegetation surveys with relevant sewage sludge application rates at the assessment sites used in this study were available for the rangeland, and surveys from other locations were useful for identifying common effects of sludge addition. Media toxicity tests were only available for agricultural systems and usually with metal salts added to sewage sludge prior to application. These tests were typically not multiple years in duration.

4.3.2.1 Single chemical toxicity data

Two methods exist for estimating toxic effects and effects thresholds of individual chemicals in soils to plants. The first is to relate concentrations of chemicals in soils to levels of phytotoxicity. The second is to relate concentrations of chemicals in plant tissues to levels of phytotoxicity. An advantage of the use of soil concentrations is the often large body of data that relates this variable to toxicity. The advantage of the use of plant tissue concentrations is that, depending on the mechanism of toxicity, these data probably represent more direct estimates of exposure of plants to contaminants than soil concentrations. The issue of bioavailability can be bypassed if plant tissue concentrations are evaluated (see Sect. 4.1.4). Tissue concentrations from hydroponic studies may be used to assess phytotoxicity. On the other hand, if the wrong tissue is measured, there may be no relationship between plant tissue concentration and toxicity.

ORNL toxicity benchmarks

Oak Ridge National Laboratory (ORNL) has developed benchmarks for screening contaminant concentrations in soils for potential risks to plants (Efroymson et al. 1997a). The phytotoxicity benchmarks were derived using growth and yield parameters, including oven-dry weights of whole plants or their parts, root length, plant height, relative growth rate, grain yield, seeds per plant, percent seed germination, and fresh and air-dry weights. These parameters were selected both because of the prevalence of growth and yield studies and because they are relevant to plant populations (as well as individuals) and to the ability of the plants to support higher trophic levels (Efroymson et al. 1997a). If measures of growth or yield were not available, studies measuring metabolic activity were occasionally used. Most data were from toxicity studies in greenhouses and growth chambers, with occasional studies undertaken in the field. The chemicals were usually added in the form of soluble salts, and sewage sludge was never the source. The bioavailability of contaminants in sewage sludge, discussed in Section 4.1.4, would be expected to influence toxicity to plants. For example, McBride (1995) asserts that free metal cations are usually more phytotoxic than soluble complexed forms of the same metals. These considerations are not explicitly incorporated into the derivation or use of ORNL phytotoxicity benchmarks Efroymson et al. 1997a).

The method used for deriving phytotoxicity benchmarks (Efroymson et al. 1997a) is based on the National Oceanographic and Atmospheric Administration's method for deriving the Effects Range Low (ER-L) (Long and Morgan, 1991), which has been recommended as a sediment screening benchmark by U.S. Environmental Protection Agency (EPA) Region IV. The ER-L is the tenth percentile of the distribution of various toxic effects thresholds for various organisms in sediments. (A threshold is defined as a 20% effects level (see Sect. 1.5.1).) Thus, the assessor using the benchmarks is 90% certain that plants growing in the site soil are protected. As with NOAA ER-Ls, statistical fitting was not used in the derivation of phytotoxicity benchmarks because there were seldom sufficient data (Efroymson et al. 1997a).

The level of confidence in the benchmarks ranges from low (e.g., molybdenum benchmark derived from 1 value in 1 study) to high (e.g., cadmium value derived from 74 values in 23 studies) (Efroymson et al. 1997a). All of the benchmarks are believed to be conservative, except for the value for molybdenum, for which the level of conservatism is unknown.

In this assessment benchmarks were first used to screen for Contaminants of Potential Ecological Concern (COPECs), i.e., to eliminate contaminants from further consideration for risks to plants if they did not exceed benchmark concentrations in soils. In addition, they provided a conservative estimate of risks from particular constituents of sewage sludge in treated soil. Both the screening of contaminants for risks and the use of phytotoxicity benchmarks in the weight of evidence are presented in the risk characterization for each ecosystem (Sect. 4.3.3).

Organic contaminants

ORNL phytotoxicity benchmarks are available for all assessment chemicals except dioxins and dibenzofurans; PCBs are the only organic chemicals in this assessment for which a benchmark was derived. Little information is available on the toxicity of dioxins to plants, probably because the uptake of the chemicals by plants is so low. For example, the uptake of TCDD by oats and soybeans from a sandy loam soil containing 0.06 mg/kg of the chemical was less than 1 μg/kg (Helling et al. 1973). Neither the soil concentration nor the plant tissue concentration has been linked to toxicity in this paper or elsewhere. Because of the low uptake, it is unlikely that TCDD at a concentration in the range of those observed in sludges in the NSSS would be a major contributor to phytotoxicity from sewage sludge. Risks to plants from TCDD are not characterized further in this assessment.

DTPA-extractable metals in soil

In the studies from which ORNL benchmarks were derived, chemical concentrations in soils were determined either by the addition of added salt concentrations to background concentrations in soil or by extraction with strong acid. As stated in Sect. 4.1.4, DTPA is an extraction method that is sometimes correlated with the availability of metals to plants. Researchers at the Rocky Mountain Forest and Range Experiment Station in Albuquerque, New Mexico, cite phytotoxicity values in Tiedemann and Lopez (1982) for DTPA-extractable metals in soil. According to these authors phytotoxicity occurs at 10 to 40 mg/kg DTPA extractable copper and >0.1 to 1.0 mg/kg DTPA-extractable cadmium.

Toxicity values for plant tissues–methods used in Technical Support Document

The Technical Support Document (EPA 1992a) evaluated risks to agricultural plants from toxicity of chromium, copper, nickel, and zinc. Chromium is not evaluated in this risk assessment. The ultimate loading limits for copper, nickel and zinc in municipal sewage sludge applied to land are based on phytotoxicity. The concentration of pollutants in plant tissue were estimated from application rates of biosolids and related to yield reduction, with a goal to establish cumulative reference application rates. An empirical relationship was derived between biosolids application rate and concentration of metal in plant tissue. The measurement endpoint was a 50% yield reduction, chosen partly because "spatial and temporal variations in crop yield are often large and may exceed 50 percent" (EPA 1992a).

Two alternative procedures were used to determine the cumulative reference application rate based on phytotoxicity, and the more conservative result was used for each metal. In the first approach, the threshold phytotoxic threshold leaf concentration of corn was established by an extensive literature review (Chang et al. 1992). Then, the probability of crops in sludge-treated soils exceeding the threshold was determined using a graph of concentration of chemical in plant tissue versus chemical loading from sludge application (kg/ha). The acceptable probability of exceedence of the threshold was set to 0.01. The choice of corn yield as a measurement endpoint was based on its wide use at sewage sludge application sites rather than its sensitivity to metal toxicity. McBride (1995) suggests that the crop is a rather metal-tolerant dicotyledonous species ; however, Efroymson et al. (1997a), who have compiled phytotoxicity data for numerous metals in soil, find that corn is not generally more tolerant than other test species, except in the case of lead.

In the second approach phytotoxic thresholds for each contaminant were determined for sensitive agricultural crops, using lettuce studies for zinc and bush and snap beans for copper toxicity (EPA 1992a). Toxicity thresholds for black-eyed susan, little bluestem, and red oak were found not to be more sensitive than sensitive agricultural plants (EPA 1992a); thus the approach was deemed valid for the protection of nonagricultural systems. (It should be noted that few plant species were represented, however.) The output of the analysis was the cumulative reference application rate (kg metal per ha), the quotient of the phytotoxicity threshold (mg metal per kg plant tissue) minus the background plant tissue concentration (mg metal per kg plant tissue) and the uptake slope of the pollutant in plant tissue ((mg metal/kg plant tissue)/(kg metal/ha)). The background concentrations of contaminants in plants were calculated by taking the geometric mean of the plant tissue concentrations in the plant group grown in unamended soils.

For zinc, approach 2 was the more conservative, with a threshold phytotoxic concentration of 400 mg/kg plant tissue (DW) and a reference cumulative application rate of 2800 kg/ha. Based on the assumptions in this assessment, this loading limit corresponds to soil concentrations ranging from 1353 mg/kg in the loblolly pine plantations to 1817 mg/kg in the Douglas-fir forest.

For copper, approach 1 was the more conservative. Data connecting leaf concentration to retardation of growth are sparse, and only two experiments were analyzed: 1) a hydroponic study with corn, where no effects occurred at 40 mg/kg plant tissue and 2) a study which showed possible toxicity at 7 mg/kg, but which authors of the TSD dismissed as probable copper deficiency (EPA 1992a). Thus the plant tissue concentration chosen for use as the 50% yield or growth reduction was 40 mg/kg. The cumulative loading rate of 1500 kg/ha was chosen, based on the lack of studies at higher concentrations. The probability that copper levels in corn exceed 40 mg/kg was less than 0.0001 at 1550 kg/ha. Based on the assumptions in this assessment, this loading limit corresponds to soil concentrations ranging from 724 mg/kg in the loblolly pine plantations to 931 mg/kg in Douglas-fir forest.

For nickel, approach 1 was the more conservative. The plant tissue concentration associated with 50% yield or growth reduction was 3 mg/kg. The observation that as the application of nickel increased, the concentration of nickel in corn on sludge-treated soil generally decreased was problematic for setting a cumulative loading rate (EPA 1992a). Between 1 and 100 kg/ha, the probability that nickel levels in corn foliage exceed 3 mg/kg is 0.0136. Between 100 and 425 kg/ha, the probability of exceedence dropped to 0.0045. The risk-

based loading rate was set to 420 kg/ha. Based on the assumptions in this assessment, this loading limit corresponds to soil concentrations ranging from 210 mg/kg in the loblolly pine plantations to 269 mg/kg in Douglas-fir forest.

The methods used in the TSD to estimate phytotoxicity (EPA 1992a) have been criticized (McBride 1995, Schmidt 1997). Although Chang et al. (1992) obtained data on numerous crop plants, they focused their effects assessment on corn. In addition to the insensitivity of corn relative to other plants (EPA 1992a and McBride 1995), the choice of the 50% yield reduction has been questioned (McBride 1995). As stated above, for this ecological risk assessment the target level of effects of concern is 20%. Moreover McBride (1995) suggests that root concentrations of heavy metals may be more appropriate indicators of incipient toxicity than plant tops; metal uptake by roots may not achieve a plateau at the same total metal content in soil as metal uptake by foliage or stems. Even Chang et al. (1992) note that leaf concentration may not be the most appropriate indicator of copper toxicity. It is also likely that the size and absorption relationship of roots and stems or foliage is different in hydroponic and soil media.

The methods used in the TSD are less conservative than the ORNL phytotoxicity benchmarks. Chang et al. (1992), the authors of Approach 1 in the TSD, acknowledge that regulators might consider altering their approach to phytotoxicity of sewage sludge through the use of a safety factor.

Toxicity values for plant tissues--derivation of benchmarks for contaminants of potential ecological concern (COPECs)

Plant tissue toxicity benchmarks were derived for all chemicals that were credible COPECs. Lowest Observed Adverse Effects Levels (LOAECs) were compiled for multiple plant species at multiple sites, and the tenth percentile was chosen as the benchmark. The distribution of LOAECs was compared to concentrations of elements measured at sewage sludge-amended sites.

4.3.2.2 Media toxicity tests of sludge-amended soils

Toxicity tests using sludge-contaminated soil have not been undertaken with soils from the ecosystems considered in this risk assessment. Also, most of the agricultural studies presented below amend sewage sludge with metal salts. The only difference between this and the single chemical toxicity line of evidence is the presence of sludge. It is possible that the studies below are among those that were used to derive phytotoxicity thresholds in the TSD.

The relative phytotoxicities of zinc, copper, and nickel to perennial ryegrass were investigated in plant pots containing sandy loam soil of pH 7.0 (Davis and Carlton-Smith 1984). Sulfates of the metals were added to sludge prior to application. The threshold concentrations in soil associated with decreased yield were: 319 mg/kg for zinc, 105 mg/kg for copper, and 221 mg/kg for nickel. These thresholds were associated with loading rates of 342, 132, and 356 kg/ha of zinc, copper, and nickel, respectively. Thresholds were determined by regression, and level of effect at the threshold (e.g., 20%) was not stated. The concentrations in foliage associated with decreased yield were: 140 mg/kg for zinc, 22 mg/kg for copper, and 90 mg/kg for nickel. Below these soil and foliage concentrations, no interactive effects were observed, but toxicities were additive above these thresholds (Davis and Carlton-Smith 1984).

Zinc added to soil as zinc sulfate-amended sewage sludge reduced the yield of lettuce at soil concentrations of 340 mg/kg for calcareous and 180 mg/kg for acid soils and of wheat at soil concentrations of 340 mg/kg for calcareous and 660 mg/kg for acid soils (Mitchell et al. 1978).

Sewage sludge containing high zinc concentrations and added to soil at the rate of 2% resulted in a level of 870 mg/kg in soil and decreased yields of red fescue (33.7% of control), soybean (16.1% of control), Romaine lettuce (1.6% of control), barley (42.8% of control). Chlorosis in the first-clipping leaves of tall fescue, perennial ryegrass, Canada bluegrass, soybean, and Romaine lettuce were also found (Chaney et al. 1989). Second-clipping shoots and other endpoints were examined in this study with toxicity to several crops at the 870 mg/kg level of zinc. Lower levels of zinc were not tested, so effects thresholds are unknown.

Smith (1994) estimated permissable levels of zinc (as well as nickel and copper) in sludge-treated soils of Harrogate and Swinton, England, based on toxic concentrations in ryegrass tissue. For zinc, the levels are 210 mg/kg for pH 5.0 to 5.5, 246 mg/kg for pH 5.5 to 6.0, and 591 mg.kg for pH > 7.0. The researcher recommends that zinc toxicity to critical microbial processes be studied (Smith 1994).

The toxicity of sewage sludge-amended soil to crop plants was tested sixteen years after application of 500 Mg/ha (Brallier et al. 1996). Crops included bush bean, cabbage, maize, lettuce, potato, and tomato. In general, the quantities of metals added in sewage sludge remained in the top 15 cm of soil. Yields of all crops except maize were significantly reduced. In additional treatments, soil was limed from a pH of 4.6 to a pH of 5.8, 6.5, or 6.9, and yields were not reduced.

4.3.2.3 Vegetation surveys

Few vegetation surveys have been undertaken on sludge-treated lands, particularly in the ecosystems of concern in this risk assessment. Where they have been conducted, many are discussed in Chapter 2, since the dominant effects on plant communities are probably attributable to the nutrients rather than heavy metal or organic contaminants in sewage sludge. Decreases in total plant density, species richness, and species diversity, increases in total plant biomass and canopy cover, and changes in abundance of particular species are discussed for particular locations in the rangeland in Sect. 2.6.

Agricultural systems

According to the TSD, crop yield reductions have been associated with sewage sludge only when it contained very high concentrations of metals or when pH was low (EPA 1992a). Copper, zinc, nickel, and chromium toxicities to even sensitive plants have not been observed in applications of sewage sludge at pH > 5.5 (W-170 1989, as cited in Chang et al. 1992).

Savannah River Plant

Liquid municipal or solid industrial sewage sludge was applied to loblolly pine plantations (establishment, 3, 8 or 27 years old) at the Savannah River Plant, Aiken, SC, with application rates determined according to nitrogen content in sewage sludge (0, 400, or 800 kg N/ha for the liquid sludge or 630 kg N/ha for the industrial sludge) (McLeod et al. 1986). Understory biomass increased for all plantations, with the herbaceous plants exhibiting the

largest increase, compared to shrub-vine and woody vegetation. The biomass peaked in the first year following application. It was suggested that increases in the biomass of perennials may lead to a consistent understory from year to year (McLeod et al. 1986).

Increased diameter and basal area of the 8- and 28-year-old trees was observed during the four years following application, though the solid sludge did not increase growth of the older trees (McKee et al. 1986). The failure of sludge to increase growth of 3-year old trees was attributed to competition from weeds.

Hubbard Brook, NH

Municipal sewage sludge from Plymouth, NH was applied to plots at the Hubbard Brook Experimental Forest at one of two application rates: 5.8 or 28 Mg/ha (Koterba et al. 1979). In the first two growing seasons following sludge application, no differences in basal area of trees were observed. On the 28 Mg/ha plot, 3 of 31 understory species showed differences in density: Maple seedlings declined by over 20% in the second summer following application, the shrub *Vibernum alnifolium* Marsh. increased 120%, and the herb *Viola pallens* Brainerd increased 40% in density.

Pack Forest understory, Washington

Sewage sludge was applied at several locations in Pack Forest, with application rates including: 1) Wyco Yellow, one application of sludge at 51 Mg/ha; 20 Wyco Green, four annual applications at 51 Mg/ha; 3) a control for 1 and 2; 4) Hugo Peak, one unevenly applied application at 300 Mg/ha; 5) Lathrop Drive, two annual applications of 51 Mg/ha; 6) HTUN (Highway Thinning-unthinned), thinned and unthinned plots with one 100 Mg/ha application followed approximately 3 years later by a 51 Mg/ha application, and thinned and unthinned control plots (Milner 1986). In all locations, the dominant plants were shrubs. The most dominant plant did not change with the sludge treatments, but at all sites but Lathrop Drive, five of the dominant species differed in relative dominance between the treatment and control plots, based on cover. These species included twin flower (*Linnea borrealis*) and snowberry (*Symphorocarpus alba*), which became less dominant in sludge-amended plots, and trailing blackberry (*Rubus vitifolius*), bracken fern (*Pteridium aquinum*), and elderberry (*Sambucus* spp.), which became more dominant. Among herbaceous plants, higher cover for many species was observed with sludge treatment, particularly at the Wyco and HTUN sites.

Community similarity indices calculated over two years ranged from 64-70% among the sludge-treated and control areas (Milner 1986). Shrub similarity coefficients ranged from 71-83%, suggesting little change with sludge treatment. Changes were greater among herbs, with similarity coefficients ranging from 43 to 67%. Increases in weeds with sludge-treatment were observed at Wyco and HTUN, probably because of physical disturbance and canopy closure associated with sludge treatment (Milner 1986).

Pack Forest trees, Washington

Tree seedlings were planted in plots where sludge was applied at either 512 Mg/ha or 204 Mg/ha (Zasoski et al. 1983). Western redcedar, black cottonwood, and coastal redwood had high mortality rates compared to control sites. The authors state that mortality of western hemlock was also increased, but the increase is not evident from the data. The increased

mortality was attributed to competition with grass and weeds. Girdling by voles, encouraged by increased biomass of grass, contributed to the mortality.

Tree nursery, WA

Seedlings of several tree species were grown in a nursery-bed containing one of the following treatments: sludge (aged in the open for 1 year in a cleared area), an aged sludge-soil mixture (sludge, subsoil, and peat at a ratio of 6:2:1), a soil-peat mixture, the soil-peat mixture fertilized with inorganic elements (N, P, K at 224, 156, 168 kg/ha, respectively), and a sawdust-sludge compost (derived from mixing sawdust and de-watered sludge on a 4:1 volume basis and aging in piles with infrequent turning for six months) (Zasoski et al. 1983). Douglas-fir, western hemlock, and Sitka spruce grew well in the sludge-soil mixture. Growth of conifers was lowest in the compost, probably because of low nitrogen availability. Western redcedar grew poorly in aged sludge, with mortality of 77% after year 1, compared to 8% for Douglas-fir and 13% for western hemlock. Mortality of western redcedar in the following treatments was 1% or less: soil-peat mixture, soil fertilized with inorganic elements, and compost. Douglas-fir showed symptoms of iron deficiency.

Vancouver Island

Municipal sewage sludge from the Greater Vancouver Regional District was applied at 69 Mg/ha (3% N) to a 10-year-old western red cedar plantation, with a salal understory (McDonald et al. 1994). One and two years after application, the height and diameter of trees was increased, relative to controls. Foliar concentrations of nitrogen, phosphorus, potassium, calcium, and magnesium were increased within a growing season of application. The salal understory appeared "luxuriant" in treated sites compared to controls; thus the trees did not obtain nutrients at the expense of salal (McDonald et al. 1994).

Maryland hardwood forest

Aerobically-digested municipal sewage sludge was applied to a hardwood forest in Maryland at rates of 0, 3, 6, and 12 Mg/ha (Aschmann et al. 1990). Two years after application, nitrogen in foliage increased with application rate in two of five tree species (dogwood (*Cornus* florida L.) and red maple (*Acer rubrum* L.)) and four of seven understory species (spicebush (*Lindera benzoin* (L.) Bl.), Virginia creeper (*Panthenocissus quinquefolia* (L.) Planch), honeysuckle (*Lonicera japonicum* Thunb., an alien species), and blackberry (*Rubus* sp.)).

Miami University Ecology Research Center near Oxford, Ohio

Milorganite, a heat-dried sewage sludge, was applied to an old field at an annual rate of 8.96 Mg/ha, divided into monthly applications between May and September. By the second year of application, sludge-treated plots were dominated by summer annuals, and control plots were dominated by perennials (Carson and Barrett 1988). At this time annuals accounted for 80% of annual net primary production in treated plots and less than 15% of annual NPP in control plots. Total net primary production was higher in sludge-treated plots than in control plots. Species diversity was higher in the control plot. Effects in sludge-treated plots were similar to those in fertilizer-treated plots; hence the effects were likely due to nutrients rather than metals in sewage sludge. During the drought of the summer of 1988, productivity of sludge-treated plots was

much more impacted than that of control plots (Bollinger et al. 1991). However, sludge-treated sites responded more quickly to the precipitation ending the drought.

Red pine plantations, Nekoosa Papers, Wisconsin

Effluent treatment plant sludge from Nekoosa Papers Wisconsin pulp and paper mills was applied at 0, 31, 63, and 94 Mg/ha to experimental plots at red pine plantations (ERT 1987). Plots were sampled at an undisclosed time following application and the presence or absence of six species of trees, eight species of grass, and eighteen species of forbs was recorded. More species were found in the treated plots (13 to 14 species) than in untreated plots (6 species). Plant cover was not dependent on treatment.

Red and white pine plantations, Michigan

An anaerobically digested municipal sludge was applied to 36-year old white pine and red pine plantations at 4.8, 9.7, and 19.3 Mg/ha (Brockway 1983). An undigested, nutrient-enriched papermill sludge was applied to 40-year-old red pine at levels up to 32 Mg/ha. Foliar nitrogen levels in pine increased where application rates were at least 16 Mg/ha. Radial growth at breast height increased with municipal sludge treatment . The biomass of understory vegetation increased up to 132% above control vegetation without any toxicity symptoms. In the first two months following application of the papermill sludge at 32 Mg/ha, understory biomass decreased, probably because of smothering from the sludge. By 14 months following application, biomass was increased on all plots (Brockway 1983).

Germany

Zasoski et al. (1983) cite two studies of sludge application to German plantations which resulted in adverse effects to the trees. Sewage sludge reduced survival of young Douglas-fir and red oak, probably because of increased pH and soil temperature and oxygen deficiency (Schwarz 1977). Sludge applied to Scots pine conferred reduced resistance to pathogens and reduced phenolic substances in trees (Huser 1977).

Montmorency County, Michigan

Indirect effects of sludge amendment to forest trees was observed in this ecosystem. Foraging elk broke stems of fertilized aspen trees (bigtooth aspen and quaking aspen) preferentially over unfertilized stems, leading to attack by *Cystospora chrysosperma*, a pathogenic fungus (Hart et al. 1986). Mortality in treated stands was 27%, compared to 4% mortality in control plots.

4.3.2.4 Effects due to modifying soil parameters

Salinity

Little information is available on the effects of change in electrical conductivity resulting from sewage sludge application on phytotoxicity. This soil parameter is only discussed in the risk characterization in reference to the semi-arid rangeland. Any effects from high salinity resulting from sewage sludge would be expected to be short-term. Electrical conductivity > 2.0 dS m^{-1} has been shown to inhibit the growth of many plants. (Bohn et al. 1979). On the other

hand Rodgers and Anderson (1995) observed a positive correlation in growth chambers between biomass of switchgrass and electrical conductivity, at least up to 10 dS m^{-1}.

Acidity

The role of pH in phytotoxicity of constituents of sewage sludge is considered in many agricultural studies. Low pH often increases the bioavailability of metals from sewage sludge and presumably also in background soil. For example, at pH below 5.5, there is an exponential increase in the solubility of aluminum and manganese, two chemicals not considered in this assessment, which can lead to phytotoxicity (EPA 1992a). Often sewage sludge is limed to reduce potential effects of pH, and in general pHs below 6 are avoided. However, limed biosolids have been shown to cause manganese deficiency in Maryland and Virginia Coastal Plain soils (Chaney 1994). pH is discussed in the risk characterization if and when it is important.

4.3.3 Risk Characterization for the Plant Community

As stated in Sect. 4.2.3, in the Risk Characterization, the results of the exposure assessment (Sect. 4.3.1) and effects assessment (Sect. 4.3.2) are integrated to estimate risks based on each line of evidence, and then a weight of evidence inference logic is applied to each assessment endpoint. Although all of these types of evidence were available for this assessment, the types of studies were not ideal. Single chemical toxicity tests from the literature (e.g., ORNL benchmarks) did not account for the effect of sewage sludge on bioavailability. Vegetation surveys were often conducted at application rates substantially different from the 40 Mg/ha assumed here, and changes occurred in the short term that may not be extrapolated to the long term or vice versa. Media toxicity tests were undertaken with agricultural plants, and often with soluble salts added to sludge.

4.3.3.1 Use of single chemical soil toxicity data for screening assessment

According to the screen against ORNL phytotoxicity benchmarks, potential risks to plants in all ecosystems may exist from exposure to zinc under the NSSS-based exposure scenario at an application rate of 40 Mg/ha. The hazard quotients for zinc are: 3.7 for the Douglas-fir forest (Table 4.41), 1.4 for the Loblolly pine plantation (Table 4.42), 2.7 for the deciduous forest (Table 4.43), and 1.8 for the semi-arid rangeland (Table 4.44).

The only other chemicals that could pose risks to plants at 40 Mg/ha, according to the screening against benchmarks, are molybdenum, with hazard quotients of 1.6 for the Douglas-fir forest (Table 4.41) and 1.7 for the deciduous forest (Table 4.43); lead, with a hazard quotient of 1.1 for the deciduous forest (Table 4.43); and selenium, with a hazard quotient of 1.13 for the rangeland (Table 4.44). The chemicals with hazard quotients above one are listed in Table 4.45.

Table 4.41 Screening of Risks to Plants from Sludge Application in Douglas-fir forest Exposure Scenario

	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	2.6	2.92	10	0.292
	Cadmium	1	1.69	4	0.423
	Copper	22	40.8	100	0.408
	Lead	45	50.0	50	1
	Mercury	0.051	0.160	0.3	0.533
	Molybdenum	3	3.29	2	1.65
	Nickel	14	16.1	30	0.535
	Selenium	0.2	0.365	1	0.365
	Zinc	120	183	50	3.66
	dioxins	0	0	NA	
	PCB-1248	0	0.0041	40	0.000102
	PCB-1254	0	0.0092	40	0.00023
	PCB-1260	0	0.0032	40	0.00008

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.42 Screening of Risks to Plants from Sludge Application in Loblolly pine plantation

Exposure Scenario	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	2	2.25	10	0.225
	Cadmium	0.2	0.743	4	0.186
	Copper	10	24.8	100	0.248
	Lead	20	23.9	50	0.478
	Mercury	0.05	0.136	0.3	0.453
	Molybdenum	1	1.23	2	0.616
	Nickel	10	11.6	30	0.387
	Selenium	0.2	0.330	1	0.330
	Zinc	20	69.6	50	1.39
	TCDD	0	8.58E-07	NA	
	PCB-1248	0	0.0032	40	0.00008
	PCB-1254	0	0.0072	40	0.00018
	PCB-1260	0	0.0025	40	0.0000625

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.43 Screening of Risks to Plants from Sludge Application in Deciduous Forest

Exposure Scenario	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	4.1	4.45	10	0.445
	Cadmium	0.2	0.960	4	0.240
	Copper	8.6	29.3	100	0.293
	Lead	48.5	54.0	50	1.08
	Mercury	0.051	0.171	0.3	0.571
	Molybdenum	3	3.32	2	1.66
	Nickel	15	17.3	30	0.575
	Selenium	0.2	0.382	1	0.382
	Zinc	61.5	131	50	2.62
	TCDD	0	1.20E0-6	NA	
	PCB-1248	0	0.0045	40	0.000113
	PCB-1254	0	0.0101	40	0.000252
	PCB-1260	0	0.0035	40	0.0000875

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.44 Screening of Risks to Plants from Sludge Application in Semi-arid Rangeland

Exposure Scenario	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	2.2	2.45	10	0.245
	Cadmium	0.55	1.09	4	0.273
	Copper	4.84	19.6	100	0.196
	Lead	12.4	16.3	50	0.326
	Mercury	0.054	0.14	0.3	0.467
	Molybdenum	0.1	0.331	2	0.166
	Nickel	11.5	13.1	30	0.437
	Selenium	1[2]	1.13	1	1.13
	Zinc	39.84	89.5	50	1.79
	TCDD	0	8.58E-07	NA	
	PCB-1248	0	0.0032	40	0.00008
	PCB-1254	0	0.0072	40	0.00018
	PCB-1260	0	0.0025	40	0.0000625

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha
[2]Value is a detection limit; actual concentration is likely to be lower (S. Loftin, pers comm., July 1998)

Table 4.45. Summary of Screening Results for Plants from Application of Sewage Sludge, 1X 40 Mg/ha (Hazard Quotients above 1)

Ecosystem	NSSS 95% UCL
Douglas-fir forest	Mo[1], Zn
Loblolly Pine Plantation	Zn
Semiarid Rangeland	Se[1], Zn
Eastern Deciduous Forest	Pb[1], Mo[1], Zn

[1] Chemical was eliminated as a COPEC in the risk characterization

The assumed background soil concentrations of zinc in the four ecosystems are: 120 mg/kg for the Douglas-fir, 61.5 mg/kg for the eastern deciduous forest, 20 for the loblolly pine plantations, and 40.7 for the semi-arid rangeland. If background levels of zinc in soil were screened using the ORNL benchmarks, potential risks would be predicted for the Douglas-fir forest (hazard quotient = 3), eastern deciduous forest (hazard quotient = 1.5), and semi-arid rangeland (hazard quotient = 1.0). Thus, assumed background concentrations of zinc contribute significantly to the hazard quotients in sewage sludge-treated soil, and whether or not there are risks to plants depends in large part on whether the background metals are mobilized in the long term. Of course, it should be noted that background concentrations of chemicals in soil are quite variable. For example, at Sierra Blanca ranch in western Texas, zinc levels vary in the top 15 cm of soil from about 33 to 76 mg/kg (David Wester, personal communication, June 1997). Thus, at some locations, the estimated hazard quotient would be less than one.

Moreover, the assumed background concentrations of molybdenum in the Douglas-fir forest and the eastern deciduous forest are both 3 mg/kg, greater than the 2 mg/kg ORNL benchmark. Sewage sludge is a relatively minor contributor of molybdenum, adding about 0.3 mg/kg, or less than ten percent of the total.

Thus, zinc, molybdenum, and lead exceeded phytotoxicity benchmarks in at least one ecosystem for at least one assessment endpoint, plant survival and production. The phytotoxicity of these three elements was further investigated in Sect. 4.3.3.1.1 to determine whether they are credible contaminants of potential ecological concern (COPECs).

4.3.3.2 Effects of retained contaminants

Zinc

ORNL soil toxicity benchmark. The ORNL benchmark for phytotoxicity from zinc is based on five studies. Three of the five show effects at zinc levels below the 183 mg/kg estimated assessment concentration in the Douglas-fir forest soil. The number of soybean seeds was reduced 28% when zinc was added as $ZnSO_4$ to a garden soil at 25 mg/kg (Aery and Sakar 1991). Lata and Veer (1990) found reductions in whole plant weight of spinach (44%) and coriander (22%) at 87 mg/kg zinc. At a pH of 5.5, soybean leaf weight was reduced by 30% when $ZnSO_4$ was added to a level of 131 mg/kg of zinc in soil (White et al. 1979).

Toxic plant tissue concentrations (Technical Support Document). Based on Approach 2 to phytotoxicity assessment in the TSD and the 15-cm depth assumption in this risk assessment, the biosolids loading limit corresponds to soil concentrations ranging from 1353 mg/kg in the loblolly pine plantations to 1817 mg/kg in the Douglas-fir forest, which is much higher than the thresholds on which the ORNL benchmarks are based and higher than the soil concentrations associated with toxicity to plants in sludge-treated soils.

Toxic plant tissue concentrations (new toxicity benchmark). For the purpose of this study, a toxicity benchmark was derived for zinc in plant tissue. Lowest observed effects concentrations (LOECs) from various toxicity studies were compiled, and the tenth percentile, 180 mg/kg, dry mass, was calculated (Table 4.46). Concentrations of zinc in plants at sewage-sludge-amended sites were also compiled (Table 4.47). If multiple application rates were tested, the rate closest to 40 Mg/ha was chosen. Concentrations of zinc in plant tissues in each ecosystem were also

Table 4.46. Toxicity data for use in deriving a plant tissue concentration benchmark for zinc.

plant	tissue LOEC (mg/kg)	tissue	effect	reference
barley (*Hordeum vulgare*)	500	root	10% reduction in root yield	Aery and Jagetiya 1997
Lygeum spartum	1350	shoot	50% reduction in plant yield	Diaz et al. 1996
barley (*Hordeum vulgare*)	522	shoot	10% reduction in shoot yield	Aery and Jagetiya 1997
onion (*Allium cepa*)	375	leaf	69% reduction in shoot yield	Gildon and Tinker 1983
onion (*Allium cepa*)	110	leaf	30% reduction in shoot yield	Gildon and Tinker 1983
white clover	1019	shoot	51% reduction in shoot yield	Gildon and Tinker 1983
sitka spruce (*Picea sitchensis*)	226[1]	foliage	estimated upper critical concentration (yield)[1]	Burton et al. 1983
lucerne (*Medicago sativa*)	215	top	66% reduction in plant yield	Carroll and Loneragen 1968
Barrel medic (*Medicago truncatula*)	374	top	89% reduction in plant yield	Carroll and Loneragen 1968
subterranean clover (*Trifolium subterraneum*)	389	top	31% reduction in plant yield	Carroll and Loneragen 1968
barley (*Hordeum vulgare*)	290[1]	shoot	estimated upper critical concentration (yield)[1]	Davis et al. 1978
upland cotton (*Gossypium hirsutum*)	200[1]	top	estimated upper critical concentration (yield)[1]	Ohki 1975
chrysanthemum (*Chrysanthemum murifolium*)	421	foliage	20% reduction in plant yield	Patel et al. 1976
bush bean (*Phaseolus vulgaris*)	321	foliage	34% reduction in plant yield	Wallace et al. 1977
soybean (*Glycine max*)	754	trifolioate leaves	69% reduction in plant yield	White et al. 1979
soybean (*Glycine max*)	614	trifolioate leaves	25% reduction in plant yield	White et al. 1979
corn (*Zea mays*)	1000	plant	20% reduction in plant yield derived from 4 studies	Chang et al. 1992
lettuce (*Latuca sativa*)	220	plant	20% reduction in plant yield	Chang et al. 1992
bush bean	100	plant	20% reduction in plant yield derived from 5 studies	Chang et al. 1992
benchmark	**180**		**10th percentile of LOECs**	

[1] The estimated upper critical concentration is generally lower than any measurable LOEC that would be statistically significant. Therefore it is a conservative estimate of the LOEC.

Table 4.47. Concentrations of zinc in plants at sewage sludge-amended sites.

ecosystem and plant type	# applications X rate (Mg/ha)	months post application	range of concns of Zn in plant (mg/kg)	reference
rangeland; herbs	1 X 40	24	16 to 41.6	Pierce 1994
rangeland; herbs, grasses; shrubs	1 X 30	12, 24	16.9 to 28.1	Harris-Pierce 1994
rangeland; grasses	1 X 45	12, 24, 60	23 to 77 *	Fresquez et al. 1991
rangeland; grasses	1 X 45	12, 24	19 to 75	Fresquez et al. 1990
Douglas-fir forest; trees	1 X 512	48	33.3 to 419.1	Greenleaf-Jenkins and Zasoski 1993
Douglas-fir forest; trees	1 X 512	24	124.5 to 1180.4	Greenleaf-Jenkins and Zasoski 1993
old field; grasses and shrub	50 X 1.792	several	41 to 97.33	Levine et al. 1989, Hall et al. 1990

estimated using the assumptions in this study (incorporation to 15-cm depth, 40 Mg/ha application rate). These concentrations were generated in Monte Carlo simulations (see Sect. 4.2.1.5), where distributions were used to represent contaminant concentrations in sewage sludge from the NSSS (Table 4.1), spatial variability in background levels of elements in surface soil (Table 4.28), and concentrations in above-ground vegetation based on the log-log regression for zinc (Table 4.6).

The distribution of LOECs, the estimated distributions of zinc in above-ground plant biomass in the four ecosystems, and the measured concentrations of zinc in plants in the rangeland 1-2 years after application of 40-45 Mg/ha were plotted (Fig. 4.8). All measured tissue concentrations of zinc in the rangeland were lower than the benchmark of 180 mg/kg. About 30-40% of the estimated concentrations in all four ecosystems were above the toxicity benchmark. Based on the criteria presented in Sect. 4.2.3.1.2, adverse effects from zinc are likely. Because measured concentrations of zinc in the rangeland are so much lower than the estimated long-term concentrations (and are lower than the benchmark), it is concluded that risks to plants in the rangeland are possible (but not probable) from zinc toxicity. Although concentrations of zinc in trees in the Douglas-fir forest exceeded the toxicity benchmark, the application rate of 512 Mg/ha is 13 times the application rate in this study, 40 Mg/ha (Table 4.47). Thus the risk estimate based on Monte Carlo simulations is used.

DTPA-extractable metals in soil. Phytotoxicity occurs at >20 to 40 mg/kg DTPA-extractable zinc (Tiedemann and Lopez 1982, as cited in Fresquez et al. 1991). At sludge applications of 45 and 90 Mg/ha in the fifth growing season following application to rangeland soil, mean DTPA-extractable zinc levels were 7.78 and 9.67 Mg/ha, respectively (Fresquez et al. 1991). Thus, no phytotoxicity of zinc would be predicted in the rangeland based on the DTPA-extractability criterion alone. In fact, prior to the addition of sludge, zinc levels in soils were less than what is recommended for plant growth (Fresquez et al. 1990). However, DTPA-extractable zinc increased substantially from the first growing season to the second and to the fifth. Zinc concentrations in rangeland soils may continue to increase for years following sludge application.

Bioavailability and mobility of zinc. Sewage sludge is not a strong barrier to the uptake of zinc by plants. The uptake of zinc has been shown to increase with increases in sludge application rates in ryegrass, (Dudka and Chlopecka 1990), fodder rape (Singh and Narwal 1984), lettuce on many soils (Hue et al. 1988), soybean (Heckman et al. 1987, Reddy and Dunn 1983), and rye, sorghum-sudan, and corn (Keeney and Walsh 1975). Levels of zinc increased linearly with application rate in bottlebrush squirreltail during the first and second growing seasons in the Rio Puerco Valley, New Mexico (Fresquez et al.1990). For the six years following the application of sewage sludge to corn plots at 60, 120, and 180 Mg/ha, zinc concentration in stover (stalks and leaves) increased linearly each year at each application rate (Bidwell and Dowdy 1987). Both acidic soils (Chaney and Ryan 1993, Valdares et al. 1983) and the presence of cadmium, at least in young lettuce leaves (McKenna et al. 1993), may lead to increased uptake of zinc.

In sludge-amended soils of the Douglas-fir forest, zinc was the most mobile metal, with the highest fraction taken up by Douglas-fir and the highest DTPA-extractable levels (Greenleaf-Jenkins and Zasoski 1986). The amount of zinc recovered by acid extraction of the top 20 cm of soil at three locations at Pack Forest, Washington after application of about 500 Mg/ha was also high, about 140%, 140%, and 56% of the estimated applied mass (Greenleaf-Jenkins and Zasoski 1986). In a loblolly pine plantation on the Oak Ridge Reservation, concentrations of zinc associated with the application of sewage sludge remained elevated in pine needles for at least a year following application (Gunderson et al. 1995).

In a field study in which sewage sludge was mixed in the top 20 cm of a California soil, the zinc content of DTPA extracts of the surface soil averaged 54% of that extracted with nitric acid for soil treated with Oakland sludge, 36% of that extracted with nitric acid for soil treated with Pacheco sludge and 22% for control plots (Williams et al.1980). It is notable that if DTPA extraction represents bioavailability, the bioavailability of zinc in sewage sludge was higher than that in the untreated soil.

In a tree nursery, DTPA-extractable zinc was monitored nine times following sewage sludge compost addition for 885 d (Korcak et al. 1979). The authors claim that extractable zinc decreased with time, and the mean extractable zinc concentration peaked within the first 78 d following sludge addition. However, at 885 d, the mean DTPA-extractable zinc concentration in soil at all application rates (56, 112, 224, and 448 Mg/ha) was higher than it had been measured at any time since 113 days following application. This evidence suggests that zinc levels in soil may increase if soils are monitored long enough.

Data do not support the elimination of zinc as a contaminant of potential ecological concern to plants in the forests, primarily because of the lack of long-term data on uptake and toxicity.

Molybdenum

Molybdenum is necessary for symbiotic nitrogen fixation by legumes and for nitrogen metabolism in all plants. Brill et al. (1987) note that molybdenum has been applied for years to legume fields to increase nitrogen fixation by *Rhizobium*. As stated in the effects assessment, the ORNL benchmark for toxicity of molybdenum is based on unspecified adverse effects on plants with the addition of 2 mg/kg molybdenum, according to a secondary source (Kabata-Pendias and Pendias 1984). Toxic threshold concentrations of the element in solution culture range from 0.5 to 9.5 mg/L. It is assumed that toxicity of molybdenum to plants in soil is not generally observed

or studied. Eisler (1985) asserts that no toxicity from molybdenum has been observed in field-grown crops.

It has been observed that alkaline soil (Chaney and Ryan 1993, Kretschmer et al. 1956) and slow plant growth rate (Kretschmer et al. 1956) are associated with high uptake of molybdenum by plants. In soils, molybdenum is absorbed by iron and aluminum hydrous oxides and by organic matter, with absorption decreasing at increased pH (Soon and Bates 1985). Molybdenum has been found to increase in plant tissues after several years of application of a lime-treated sludge (Soon and Bates 1985).

In Wolcott, CO, a representative rangeland site, molybdenum was observed to decrease with increasing sludge application in two of three dominant forage grasses, perhaps because of preferential uptake of copper related to nitrogen metabolism (Pierce 1994). However, the element was measured in foliage, and Wallace and Romney (1977) claim that the roots are usually the major reservoir of molybdenum.

Little information is available on the toxicity of molybdenum to plants to support or refute the screening assessment. Because the hazard quotients for molybdenum in both the eastern deciduous forest and the Douglas-fir forest are less than two, neither system is very alkaline, and plants in the ecosystems are presumably acclimated to the relatively high background levels of molybdenum, molybdenum is not a COPEC, at least with respect to plants at the 40 Mg/ha application rate.

Lead

The hazard quotient for lead in the deciduous forest is not above 1 if appropriate significant figures are used, given that the ORNL phytotoxicity benchmark has only one significant digit. In addition, the application of municipal sewage sludge to soil frequently reduces the uptake of lead by plants, probably because of added hydrous iron oxides, carbonates, and organic matter (Chaney 1990, Chaney 1994). To retain lead as a COPEC, it would have to be demonstrated that lead is as bioavailable to plants as the lead salts in the test soils from which the ORNL phytotoxicity benchmark was derived. That is not likely to be the case. Therefore, lead is not a contaminant of potential ecological concern, at least with respect to plants.

Selenium

The hazard quotient for selenium in the rangeland is not above 1 if appropriate significant figures are used, given that the ORNL phytotoxicity benchmark has only one significant digit. In addition, a detection limit of 1 mg/kg (S. Loftin, pers comm) was used as the background concentration for selenium in the rangeland. Background soil concentrations are rarely that high. Therefore, selenium is not a contaminant of potential ecological concern, at least with respect to plants.

4.3.3.3 Effects of DTPA-extractable chemicals in rangeland

As stated in the effects assessment, phytotoxicity occurs at 10 to 40 mg/kg DTPA-extractable copper and >0.1 to 1.0 mg/kg DTPA-extractable cadmium (Tiedemann and Lopez 1982, as cited in Fresquez et al. 1991). In the Rio Puerco Valley, New Mexico, DTPA-extractable copper and cadmium increased to levels considered phytotoxic when sludge was

applied at 45 and 90 Mg/ha, causing a slight drop in pH (Fresquez et al. 1991, Aguilar et al. 1994a). The mean DTPA-extractable soil concentrations of copper for the 45 and 90 Mg/ha application rates were 23.52 and 29.78 mg/kg, respectively. For cadmium, the concentrations were 0.15 and 0.20 mg/kg. Results concerning zinc are described above (Sect. 4.3.3.1.1). These concentrations, taken after the fifth growing season following application, were an order of magnitude higher than those measured after two growing seasons. (Prior to application, copper levels were lower than what is recommended for good plant growth (Fresquez et al. 1990)). Concentrations of metals were not reported after the fifth season following application.

4.3.3.4 Media toxicity tests

Ideally, the media toxicity test line of evidence would be a comparison of the phytotoxicity of sludge-treated soils (at 40 Mg/ha) with that of untreated soils from each of the four ecosystems. Instead, the available toxicity tests are of soils treated with metal-spiked sludges from agricultural systems. Since zinc is the only COPEC, only zinc-spiked soils are considered here. Toxicity tests with zinc added to sewage sludge had phytotoxicity thresholds ranging from 180 to 870 mg/kg of zinc in soil. The 180 mg/kg level is less than the predicted soil level of zinc in the Douglas-fir forest (183 mg/kg) at a 40 Mg/ha application rate, using the 95% UCL. The 180 mg/kg zinc level was from lettuce in a pH 5.7 soil (Mitchell et al. 1978), compared to the pH levels of 5.0, 5.4, and 4.6 in regions of Pack Forest (Greenleaf-Jenkins and Zasoski 1986a).

The toxicity test evidence suggests that zinc in sewage sludge may pose a risk to plants in the Douglas-fir forest. Zinc is sewage sludge does not pose a risk to deciduous forest, pine plantations, or semi-arid rangeland, based on the media toxicity tests.

4.3.3.5 Vegetation surveys

The often complex changes in vegetation with sludge application are detailed in Sect. 4.3.2.3 (forests) and in Sect 2.6 (rangeland). Effects observed or expected at the 40 Mg/ha assessment application rate are presented in the weight-of-evidence tables.

Semi-arid rangeland

The risks to or changes in plants in the semi-arid rangeland that are relevant to the assumed 40 Mg/ha application rate are those that were observed to be associated with single applications of sewage sludge between 30 and 45 Mg/ha (Sect. 2.6). Substantial changes in vegetation and in parameters such as biomass and cover were often observed. Nutrient contents of plants increased, organic matter in soil sometimes increased, species richness and diversity sometimes decreased, total biomass and canopy cover usually increased, root growth decreased in one case, and the density of a noxious plant decreased. Whether or not these constitute risks depends on the regulatory and site-specific management goals.

An additional hypothesized interaction of nitrogen and low precipitation may result in adverse effects to plants in the rangeland. Sludge loading rates as low as 22.5 Mg/ha provide a large increase in nitrogen. Sludge application is often associated with increased plant growth during periods of high water availability. However, the high-yield plants on sludge-amended sites are more susceptible to summer drought than plants on untreated sites (S. Loftin, pers. comm. May 1997).

Understory of forests

Increased biomass and changes in species density with sludge application have been observed in forest understories. Based on the plant community or species density changes in Pack Forest, Hubbard Brook Experimental Forest, the Savannah River Plant, the semi-arid rangeland (Sect. 2.6) and the Miami of Ohio old field research site, it is likely that the herbaceous (and perhaps the woody) understory of the Douglas-fir, eastern deciduous, and loblolly pine forests would be similarly altered with sludge amendment at the 40 Mg/ha rate.

Forest trees

Increases in biomass and/or basal area are common in sludge-treated forests.

4.3.3.6 Modeling of production

Long-term, predicted changes in biomass and community structure of the forests are presented in Chapter 2. Relative densities of different tree species may change with the application of sewage sludge, as in the modeling of forest production in the Hubbard Brook Experimental Forest Sect. 2.4.3). These results do not need to be repeated here, but they do contribute to the weight of evidence, which considers risks and changes from nutrients in sewage sludge in addition to contaminants.

4.3.3.7 Salinity in the rangeland

Increases in salts may lead to stress in plants at electrical conductivity values of 2 dS m^{-1}. Electrical conductivity, or increased soil salt concentrations, were associated with sludge application six months following treatment at the Sevilleta National Wildlife Refuge (Aguilar et al. 1994b). Although Aguilar et al. (1994b) suggest that effects should be short-lived because of the movement of the salts downward in the soil profile, high electrical conductivities (up to 3 dS/m) were measured in the top 15 cm of soil 1.5 years after application. Fresquez et al. (1990) measured electrical conductivities of 0.36, 1.06, 1.66, and 2.23 dS m^{-1} for applications of sewage sludge at 0, 22.5, 45, and 90 Mg/ha in the first growing season following application. After the second growing season, electrical conductivities were reduced to 75 to 91% of these values, with electrical conductivity at the 90 Mg/ha application reduced below the critical 2 dS m^{-1}.

4.3.4 Weight of Evidence

The weights of evidence for the assessment endpoints of the plant community in the Douglas-fir forest, loblolly pine plantation, eastern deciduous forest, and southwestern semi-arid rangeland are presented in Tables 4.48 through 4.51. Sewage sludge may present a hazard to plant growth and survival in the four ecosystems, based on zinc toxicity. Zinc added in sewage sludge may become increasingly bioavailable to plants in the long term; little evidence disputes this contention. Sewage sludge may also present a hazard to plant growth and survival in the rangeland ecosystem, based on copper and/or cadmium toxicity. The evidence is somewhat ambiguous in all ecosystems because of the lack of long-term empirical data; thus they receive a "±" estimate of risk for the endpoint of individual plant growth and survival. Little weight was given to the soil loading limits from the TSD because of the criticism of these methods (e.g., choice of 50% effects level, reliance on corn) and because the metal concentrations in soil

associated with soil loading limits are so much higher than the ORNL benchmarks or the media toxicity test thresholds.

Herbaceous community composition and biomass are likely to change in forests, but whether the change constitutes a risk is dependent on the management goals for the forest or plantation. For example, if managed for timber or paper pulp, the understory may only be important in relation to nutrient processes and water balance, as they affect the trees. Similarly, some of the observed changes in the rangeland may be viewed as positive if the ecosystem is managed for grazing land (e.g., increased plant biomass, increased foliar nutrients, potential reduced density of noxious species). Modeling is useful in predicting the increased biomass and production, but this effect would not likely be deemed an adverse effect under any management scenario for the three ecosystems. If the predicted change in overstory composition in the deciduous forest is considered undesirable, then this endpoint would also be termed at risk.

Table 4.48. Weight of evidence for plant communities in the Douglas-fir forest, 1 X 40 Mg/ha application

Endpoint	Evidence	Result	Effect or Risk[1]	Explanation
Reduction in growth, yield or survival of plant populations	Single Chemical Toxicity Data (ORNL benchmarks)	+		Based upon comparison of estimated contaminant exposures to literature-derived toxicity data, Zn presents a hazard to individual plants within sludge-treated areas.
	Single Chemical Toxicity Data (TSD)	–		Based upon methods used in the TSD to predict phytotoxicity of cadmium, chromium, and zinc, the levels applied in this scenario are not expected to result in toxic effects.
	Single Chemical Toxicity Data (new tissue benchmark for Zn)	+		About 35% of the distribution of estimated concentrations of zinc in above-ground vegetation is within the toxic range. Measured concentrations of zinc in Douglas-fir at sludge-amended sites are within the toxic range, but application rates are much higher than those in this study.
	Toxicity tests	±		Toxicity to plants from zinc added to sewage sludge has been observed at soil concentrations as low as 180 mg/kg in an acidic soil, but toxicity thresholds are usually higher.
	Weight of Evidence	+	Risk	Zinc may present a risk to plant populations.
Change in herbaceous biomass	Vegetation survey	+	Effect	Biomass of salal increased significantly with sludge application in western red cedar plantations on Vancouver Island.
Change in understory composition	Vegetation survey	+	Effect	Based on the changes in shrub and herb dominance and increased biomass in Pack Forest and at other sites, it is likely that the herbaceous understory would be altered with sludge amendment.
Increase in production of dominant forest tree (Douglas-fir)	Model results	+	Effect	Douglas-fir showed substantial increases in aboveground growth for 100 years of simulation. Application of sludge almost doubled the NPP at 50 and 100 years post-application.

[1] *Risk* indicates an adverse effect; *effect* indicates a change in the community that may or may not be considered adverse.

Table 4.49. Weight of evidence for plant communities in the loblolly pine plantation, 1 X 40 Mg/ha application

Endpoint	Evidence	Result	Risk or Effect[1]	Explanation
Reduction in growth, yield or survival of plant populations	Single Chemical Toxicity Data (ORNL benchmarks)	+		Based upon comparison of estimated contaminant exposures to literature-derived toxicity data, Zn presents a hazard to individual plants within sludge-treated areas.
	Single Chemical Toxicity Data (TSD)	–		Based upon methods used in the TSD to predict phytotoxicity of cadmium, chromium, and zinc, the levels applied in this scenario are not expected to result in toxic effects.
	Single Chemical Toxicity Data (tissue benchmark for Zn)	+		About 40% of the distribution of estimated concentrations of zinc in above-ground vegetation is within the toxic range.
	Toxicity tests	–		Toxicity to plants from zinc added to sewage sludge has not been observed at soil concentrations as low as the 69.6 mg/kg in soil.
	Weight of Evidence	±	Risk	Zinc may present a risk to plant populations.
Change in herbaceous biomass	Vegetation survey	+	Effect	Understory biomass increased following sludge application at the Savannah River Plant. The increase was particularly large for the herbaceous understory.
Increase in production of dominant forest tree (loblolly pine)	Vegetation survey	+		Increases in basal area from sludge application at the Savannah River Plant with sludge application were observed, though 3-year-old trees did not display increased growth, probably because of competition from weeds.
	Model results	+		Loblolly pine showed substantial increases in aboveground growth for 50 years of simulation; the increase was evident after 15 years. Application of sludge increased the NPP by 50% at 50 and 100 years.
	Weight of evidence	+	Effect	Increased production of loblolly pine is anticipated from the application of sewage sludge.

[1] *Risk* indicates an adverse effect; *effect* indicates a change in the community that may or may not be considered adverse.

Table 4.50. Weight of evidence for plant communities in the eastern deciduous forest, 1 X 40 Mg/ha application

Endpoint	Evidence	Result	Risk or Effect[1]	Explanation
Reduction in growth, yield, or survival of plant populations	Single Chemical Toxicity Data (ORNL benchmarks)	+		Based upon comparison of estimated contaminant exposures to literature-derived toxicity data, Zn presents a hazard to individual plants within sludge-treated areas.
	Single Chemical Toxicity Data (TSD)	–		Based upon methods used in the TSD to predict phytotoxicity of cadmium, chromium, and zinc, the levels applied in this scenario are not expected to result in toxic effects.
	Single Chemical Toxicity Data (tissue benchmark for Zn)	+		About 40% of the distribution of estimated concentrations of zinc in above-ground vegetation is within the toxic range.
	Toxicity tests	–		Toxicity to plants from zinc added to sewage sludge has not been observed at soil concentrations as low as the 61.5 mg/kg in soil.
	Weight of Evidence	±	Risk	Zinc may present a risk to plant populations.
Change in understory composition	Vegetation survey	+	Effect	Sludge application at Hubbard Brook resulted in altered densities of three of 31 understory species.
Increase in nutrient content of plants	Vegetation survey	+	Effect	In a hardwood forest in Maryland, nitrogen in foliage increased in two of five tree species and four of seven understory species.
Increase in production of trees	Vegetation survey	–		No increase in basal area of trees was observed for two years following empirical applications of sludge at Hubbard Brook.
	Model results	+		Total biomass of trees at Hubbard Brook increased slightly with sludge application, peaking at year 200 (out of 500) at about 15% over the untreated plot. Application of sewage sludge slightly increased NPP over the no-treatment simulation.
	Weight of	+	Effect	A small increase in forest production is

	evidence			anticipated from the application of sewage sludge to deciduous forests.
Change in relative dominance of forest trees	Model results	+	Effect	Growth of yellow birch, a dominant species at Hubbard Brook, was enhanced with biosolids application. Sugar maple and beech were largely unaffected. Of the minor species, the biomass of red spruce was predicted to decrease and pin cherry and mountain-ash to decrease in the 100 years following sludge application.

[1] *Risk* indicates an adverse effect; *effect* indicates a change in the community that may or may not be considered adverse.

Table 4.51. Weight of evidence for plant communities in the semi-arid rangeland, 1 X 40 Mg/ha application

Endpoint	Evidence	Result	Risk or Effect[1]	Explanation
Reduction in growth, yield, or survival of plant populations	Single Chemical Toxicity Data (ORNL benchmarks)	+		Based upon comparison of estimated contaminant exposures to literature-derived toxicity data, Zn presents a hazard to plant populations within sludge-treated areas.
	Single Chemical Toxicity Data (DTPA-extractable concns)	+		Based upon DTPA-extractable concentrations of Cu and Cd in soil, these elements may pose a hazard to populations of plants within sludge-treated areas.
	Single Chemical Toxicity Data (TSD)	–		Based upon methods used in the TSD to predict phytotoxicity of cadmium, chromium, and zinc, the levels applied in this scenario are not expected to result in toxic effects.
	Single Chemical Toxicity Data (new tissue benchmark for Zn)	±		Concentrations of zinc measured in plant foliage following (1-2 years) sludge application in Colorado and New Mexico are below the toxic range. About 30% of the distribution of estimated concentrations of zinc in above-ground vegetation is within the toxic range.
	Electrical conductivity Data	+		Levels of salinity may be toxic to plants in Rio Puerco Valley, New Mexico, at least in the short term.
	Toxicity tests	–		Toxicity to plants from zinc added to sewage sludge has not been observed at soil concentrations as low as the 40.7 mg/kg in soil.

	Biological surveys	+		Increases in biomass that are associated with nutrients in sludge are sometimes followed by increased plant mortality during droughts.
	Weight of Evidence	±	Risk	Risks to plant populations may exist from zinc, cadmium, copper, salinity, or the combined stress of nutrient growth surges and drought.
Increase in nutrient content of plants	Vegetation survey	–	Effect	Nitrogen, phosphorus, potassium, and sometimes protein content of plants increased with sludge application.
Change in species richness and diversity	Vegetation survey	+	Effect	Species richness and diversity sometimes decreased with sludge application, while the biomass of noxious broom snakeweed decreased.
Change in plant community biomass	Vegetation survey	+	Effect	Total biomass and canopy cover usually increased with sludge application.

[1] *Risk* indicates an adverse effect; *effect* indicates a change in the community that may or may not be considered adverse.

4.3.5 Uncertainties

- **Background concentrations.** Background concentrations of chemicals in soil were often responsible for a substantial fraction of the estimated toxicity in sludge-treated ecosystems. Background levels vary significantly among the ecosystems in this study. The spatial variability of background levels within an ecosystem and that within sludge-treated ecosystems in the United States have not been incorporated in this portion of the assessment. The likelihood of mobilization of background levels of chemicals by the application of sewage sludge is unknown.

- **Incorporation of sewage sludge.** The rate of incorporation of chemicals in sewage sludge into soils is unknown, as is the conservatism of the assumption that the sewage sludge is mixed in the surface 15 cm of soil.

- **Bioavailability of Contaminants.** The potential difference in bioavailability between chemicals in sewage sludge-treated soil and chemicals in untreated soil has been discussed previously. It is also notable that bioavailability may vary from sludge to sludge and from soil to soil. Toxicity tests rarely came from the particular assessment ecosystems for which they were used.

- **Decomposition of sludge and bioavailability of chemicals.** The bioavailability of metals and other chemicals applied to soils in sewage sludge may be slowly altered with the decomposition of organic matter in sludge.

- **Extrapolation of ambient media toxicity test data.** It has been assumed that media toxicity tests and biota surveys at application rates of 40 Mg/ha are comparable to the assumptions in this assessment. To determine if this is true, the concentrations of active chemicals (particular nutrients or contaminants) in sludge would have to be measured, as well as the ultimate concentrations in soil.

- **Extrapolation from Published Plant Toxicity Data**. Published toxicity studies emphasize crops rather than forest or rangeland species. In addition, soils are not generally representative of the soils in the assessment ecosystems. Toxicity thresholds for plants in assessment ecosystems are likely to vary from those in published studies.

- **Single Contaminant Tests vs Exposure to Multiple Contaminants in the Field.** Although organisms in the four ecosystems are exposed to multiple contaminants concurrently, published toxicological values from which ORNL benchmarks were derived only consider effects experienced from exposures to single contaminants. Because some contaminants to which wildlife and plants are exposed can interact antagonistically, single contaminant studies may overestimate their toxic potential. Similarly, for those contaminants that interact additively or synergistically, single contaminant studies may underestimate their toxic potential.

- **Inorganic Forms or Species Present in the Environment.** Toxicity of metal species varies dramatically depending upon the valence state or form (organic or inorganic) of the metal. Relative toxicities to plants are generally unknown. The NSSS does not report species or form of contaminants in municipal sewage sludges.

4.4 RISKS TO SOIL INVERTEBRATES

4.4.1 Exposure Assessment for Soil Invertebrates

The general exposure assumptions for the ecological risk assessment are stated in Section 4.1. A critical assumption for soil invertebrates is that constituents of sludge are evenly distributed through the top 15 cm of soil. For the purposes of this study, it is assumed that earthworms and other invertebrates are exposed entirely to these soils rather than partly to soils beneath or primarily to the litter layer or unincorporated sewage sludge. In reality, mites and other arthropods that inhabit the litter layer may be exposed primarily to the surface soil and litter. The soil concentrations of contaminants to which the invertebrates are exposed are presented in Tables 4.2, Douglas-fir forest; 4.3, loblolly pine plantations; 4.4, deciduous forest; and 4.5, semi-arid rangeland.

It is assumed that earthworms represent highly exposed soil invertebrates, though these may not be the most sensitive organisms to chemicals in sewage sludge-treated soil. These organisms absorb a variety of inorganic and organic soil contaminants through both feeding in soil and litter and burrowing in soil. Earthworms may be exposed through oral ingestion or through the cuticle. Exposures are different for organisms that feed on well-decomposed material near the surface, such as *Eisenia fetida*, the most common worm used in toxicity tests, and those that pull litter into burrows in the subsoil. Earthworms of the species *Lumbricus terrestris* have been found deep in the soil in December and in the upper layers in June (Andersen 1979). The worms are sluggish in December and active in June. Moreover, Andersen (1979) found that *Allolobophora* species have taken up more lead in the autumn than in the spring, with the difference related to level of activity.

Earthworms are not common in the semi-arid rangeland (or in water-logged soils in forests). Since the majority of toxicity information regarding soil invertebrates is based on earthworm data, the assessment of risks to soil invertebrates has limited utility for organisms inhabiting the semi-arid rangeland.

Many insects are exposed to metals from sewage sludge primarily through plants. These exposures are considered in the section for the sake of a complete review and because sewage sludge has been demonstrated to impact some of these organisms. However, herbivorous insects are not an assessment endpoint.

4.4.2 Effects Assessment for Soil Invertebrates

In an ideal ecological risk assessment, three lines of evidence are weighed: single chemical toxicity data, biological surveys of the affected system, and toxicity tests with ambient media, in this case sewage sludge-treated soils or sewage sludge alone. Single-chemical toxicity data were available for most contaminants in sludge, usually from experiments with metal salts. Media toxicity tests were not performed in the assessment ecosystems, and few toxicity tests with sewage sludge and earthworms of other invertebrates were available. Surveys of invertebrates were not undertaken in the assessment ecosystems, and those that were performed often used liquid sludge effluent, which is not very relevant to this study. Invertebrate surveys and toxicity tests were typically not multiple years in duration.

4.4.2.1 Single chemical soil toxicity

ORNL benchmarks

Oak Ridge National Laboratory (ORNL) has developed benchmarks for screening contaminant concentrations in soils for potential risks to soil invertebrates (Efroymson et al. 1997b). The benchmarks were derived from studies of survivorship, growth, and reproduction (cocoon production, cocoon hatching rates) of earthworms. Where survival was less than 50%, the associated concentration in soil was divided by 5 to approximate the ratio LC_{50}/EC_{20} (Efroymson et al. 1997b). Little is known about the mechanisms of toxicity to earthworms (Efroymson et al. 1997b). Most data used to derive the benchmarks were from toxicity studies in the laboratory. The metals were often added in the form of soluble salts, and sewage sludge was never the source. The bioavailability of contaminants in sewage sludge is discussed in Section 4.1.4.

The method used for deriving benchmarks for toxicity to soil invertebrates (Efroymson et al. 1997b) is based on the National Oceanographic and Atmospheric Administration's method for deriving the Effects Range Low (ER-L) (Long and Morgan, 1991), which has been recommended as a sediment screening benchmark by U.S. Environmental Protection Agency (EPA) Region IV. The ER-L is the tenth percentile of the distribution of various toxic effects thresholds for various organisms in sediments. (A threshold is defined as a 20% effects level (see Sect. 1.5.1).) Thus, the assessor using the benchmarks is 90% certain that soil invertebrates growing in the site soil are protected. As with NOAA ER-Ls, statistical fitting was not used in the derivation of benchmarks because there were seldom sufficient data (Efroymson et al. 1997b).

The level of confidence in the benchmarks ranges from low (e.g., arsenic and selenium benchmark each derived from 1 value in 1 study) to moderate (e.g., cadmium value derived from 16 values in 8 studies) (Efroymson et al. 1997b). All of the benchmarks are conservative, except for the values for arsenic and selenium, for which the conservatism is unknown.

Benchmarks were used first to screen for Contaminants of Potential Ecological Concern, i.e., to eliminate contaminants from further consideration for risks to soil invertebrates if they did not exceed benchmark concentrations in soils, and to provide a conservative (if somewhat poor) estimate of risks from particular constituents of sewage sludge. Both the screening of contaminants for risks and the use of phytotoxicity benchmarks in the weight of evidence are presented in the risk characterization for each ecosystem (Sect. 4.4.3).

Other single-chemical toxicity data

Molybdenum. An ORNL toxicity benchmark for toxicity of molybdenum to soil invertebrates has not been derived. Brill et al. (1987) found that survival of the termite *Reticulitermes flavipes* after 48 days of exposure to baits containing molybdenum at 1000 mg/kg was less than one percent. Bacteria in termite guts fix nitrogen, and molybdenum is required for the nitrogenase enzyme. Several species of beetles and cockroaches were not affected by exposures to 5000 mg/kg in baits.

Dioxin. An ORNL toxicity benchmark for toxicity of dioxin to soil invertebrates has not been derived. Earthworms of the species *Allolobophora caliginosa* were exposed to concentrations up to 10 mg/kg in soil (Reinecke and Nash 1984). At 5 mg/kg, survival was 100% after 85 days. At 10 mg/kg none of the worms survived after day 30. Thus the threshold for toxicity to these earthworms is between 5 and 10 mg/kg.

Toxicity of copper to earthworms (Technical Support Document)

Based on an initial screening before the TSD was written, it was determined that soil organisms were potentially at risk from copper but from no other contaminants in sewage sludge (EPA 1992a). The analysis in the TSD used a single reference, Hartenstein et al. (1980), to justify a NOAEL of 1500 mg/kg soil. The experiment is described in Sect. 4.4.2.2. *Eisenia foetida* were grown in sewage sludge containing 1500 mg/kg copper, with no apparent adverse effects. From this NOAEL the reference cumulative application rate of copper, 2900 kg/ha, was calculated (EPA 1992a).

4.4.2.2 Media toxicity tests

Media toxicity tests for soil invertebrates have not been performed at application sites where results are directly applicable to the four ecosystems in this risk assessment. Although the following studies are not ideal evidence for this assessment, the data are useful.

Sludge in soil cores, earthworms

Fifty grams of wet sewage sludge (20% solids dry weight) was added to cylinders containing about 200 cm^3 of wet surface soil from a mixed northern hardwood forest in Lafayette Experiment Station, Syracuse, NY (Hamilton et al. 1988). One adult *Lumbricus terrestris* or two adult *Eisenia fetida* were added to treated and unamended soil in the cylinders for a period of 46 days. Some treatments contained both species. Both species increased in biomass in the sludge-amended microcosms, though *L. terrestris* did not increase in weight if *E. fetida* was present. The population density of nematodes was reduced in sludge-treated microcosms containing earthworms.

Aerobic sludge without soil, earthworms

Hartenstein et al. (1981) used dewatered sewage sludge from Onondaga County, New York, to test the toxicity of constituents of sewage sludge to *Eisenia foetida*. Metal salts or oxides were added to the sludge in the absence of soil. The threshold concentrations of elements in sludge that would inhibit growth of *Eisenia foetida*. were: mercury, 480-4800 mg/kg; copper, 1100-11000 mg/kg; nickel, 1200-12000 mg/kg, zinc, 1300-13000 mg/kg, and cadmium, 1800-18000 mg/kg. Mercury was the most lethal metal of those tested, but the concentrations usually found in sewage sludge were not sufficient to kill this earthworm species. The authors speculated that microorganisms which take up metals may not all be digestible by *Eisenia foetida* (Hartenstein et al. 1981).

Acid mine spoil amended with sewage sludge, earthworms

Acid mine spoil from an abandoned coal mine in Pennsylvania was mixed with sewage sludge at a rate equivalent to 50 Mg sludge/ha of mine spoil (Pallant and Hilster 1996). Treatments included with and without earthworms (3 *Lumbricus terrestris* per 3-l pot), with and without sludge, and pH 5.3 or limed to pH 6.5. Pots were planted with two grasses used in the reclamation of mine spoils: timothy and tall fescue. Earthworm survival after 10 weeks was not significantly different among the treatments. Earthworms in the sludge-amended mine spoil treatments increased in mean mass by 24% in the pH 5.3 treatment and 31% in the pH 6.5

treatment. Earthworms added to treatments containing spoils without sludge decreased in mass an average of 20%.

4.4.2.3 Surveys of soil invertebrates

Invertebrate surveys were available for the loblolly pine plantation at the Savannah River plant and the eastern deciduous forest (Syracuse, NY). In addition, invertebrate surveys from other sites are presented.

Savannah River Plant, South Carolina

Eight-year-old and twenty-eight-year-old loblolly pine stands, to which sludge from Augusta, GA or Aiken, SC was applied at the Savannah River Plant, were sampled for soil mesofauna (MacConnell et al. 1986). The Augusta, GA sewage sludge, applied at 400 and 800 kg of nitrogen per hectare, reduced the number of animals present. The depth of sampling was not stated. The 630 kg N/ha of Aiken, SC sewage sludge, applied only to the 28-year-old plantation, increased the mesofaunal populations over control populations. The Aiken sewage sludge was less decomposed and had a higher solids content than the sludge from Augusta, GA. The number of mites decreased with the application of Augusta sludge, but the decrease was not necessarily statistically significant. The number of mites and collembola (springtails) increased when Aiken, SC sludge was applied at 630 kg N/ha (MacConnell et al. 1986). The authors note that, "the Augusta sludge coated vegetative matter and puddled on the forest floor, while the Aiken sludge formed aggregates or irregular shapes, ranging in size up to several centimeters in diameter." The Augusta sludge may have resulted in soil clogging from bacteria using the higher nitrogen content of that sludge (MacConnell et al. 1986).

Lafayette Experimental Station, Syracuse, NY

Activated sludge was applied at 56 Mg/ha (dry) to 4m^2 plots at the Lafayette Experimental Station. The plots were in a mixed, northern-hardwood forest, a 40-year old Norway spruce (*Picea abies* L.) plantation, and a grassy old field (Hamilton and Dindal 1989). Two plots in each ecosystem were treated with sludge, and two served as reference plots. 1600 *Eisenia fetida* per m^2 were added to one sludge-amended and one non-amended plot at each location. After six months in the mixed hardwood and old field sites, sewage sludge treatment increased both individual biomass and density of adult earthworms (*Lumbricus terrestris*). At the Norway spruce site, sewage sludge amendments decreased the density of the earthworms while increasing individual biomass. Unlike the other sites the Norway spruce plantation was very shaded throughout the study, and sludge persisted in a 5 cm layer on the soil surface, probably leading to anaerobic soil conditions. In contrast, at the other two sites, most of the sludge was incorporated rapidly into soil.

In most treatments in which *Eisenia fetida* were added, the introduced earthworms decreased in density during the 1-year study (Hamilton and Dindal 1989). An exception was in the Norway spruce plantation, where the lack of a decrease may be attributed to the shaded conditions and moist sludge layer. The introduction of *Eisenia fetida* decreased the density of *Lumbricus terrestris* in the hardwood plots.

In a separate study sewage sludge was applied at 1000 Mg/ha (wet) to an old field, a hardwood forest, and a Norway spruce plantation (Stevenson et al. 1984). Population densities

of nematodes increased with sludge addition, and population densities of enchytraeids increased in the spring only.

Red pine plantations, Nekoosa Papers, WI

Effluent treatment plant sludge from the Nekoosa Papers Wisconsin pulp and paper mills was applied at an average rate of 49 Mg/ha to experimental plots at red pine plantations (ERT 1987). Plots were sampled for earthworms, litter invertebrates, and soil invertebrates seven to nine months following application. Treatment plots had more than five times the biomass of earthworms as control locations. Annelids were frequently observed at the soil-sludge interface, but in control plots they were infrequently observed in the litter. Arthropod densities were not statistically significantly different between control and treatment plots. Within the soil, treatment with sludge was associated with lower arthropod taxa (2.8 per 242 cm^2 plot) than the control plots (3.8 per plot). Mites were dominant in control plots, but springtails and thrips were also important in sludge-amended plots.

Denmark

Earthworms of the species *Allolobophora longa, A. Caliginosa, A. rosea*, and *Lumbricus terrestris* were sampled from fields at the Askov Experiment Station treated with 30 Mg/ha (sandfree) sewage sludge (Andersen 1979). Numbers and biomass per 1/2 ha were recorded. The overall numbers and biomass of earthworms, number of *A. longa* and the biomass of *L. terrestris* increased significantly over potassium-fertilized soils.

4.4.3 Risk Characterization for Soil Invertebrates

As stated in Sect. 4.2.3, in Risk Characterization, the results of the exposure assessment (Sect. 4.4.1) and effects assessment (Sect. 4.4.2) are integrated to estimate risks based on each line of evidence, and then a weight of evidence inference logic is applied to each assessment endpoint. Although three types of evidence were available for this assessment, the types of studies were not ideal. Single-chemical toxicity tests from the literature (i.e., ORNL benchmarks) did not account for the effect of sewage sludge on bioavailability. Although this line of evidence was used to screen for potential effects in the rangeland, the earthworm-derived benchmarks are not appropriate for assessing risks to soil invertebrates in an ecosystem where there are few earthworms. In addition, surveys of invertebrates were sometimes conducted at application rates substantially different from the 40 Mg/ha assumed here, often using sludge that was not dewatered, and changes occurred in the short term that may not be extrapolated to the long term. Moreover, few media toxicity tests were undertaken.

4.4.3.1 Single-chemical soil toxicity data

According to the screen against ORNL benchmarks, potential risks to soil invertebrates in all ecosystems may exist from exposure to mercury under the NSSS-based exposure scenario at an application rate of 40 Mg/ha. The hazard quotients for mercury are: 1.6 for the Douglas-fir forest (Table 4.52), 1.4 for the Loblolly pine plantation (Table 4.53), 1.7 for the deciduous forest (Table 4.54), and 1.4 for the semi-arid rangeland (Table 4.55). All of these quotients exceed one (Table 4.56).

Table 4.52 Screening of Risks to Soil Invertebrates from Sludge Application in Douglas-fir forest

Exposure Scenario	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	2.6	2.92	60	0.0486
	Cadmium	1	1.69	20	0.0845
	Copper	22	40.8	60	0.680
	Lead	45	50.0	500	0.100
	Mercury	0.051	0.160	0.1	1.60
	Molybdenum	3	3.29	NA	
	Nickel	14	16.1	200	0.0803
	Selenium	0.2	0.365	70	0.00522
	Zinc	120	183	200	0.916
	chlorinated dioxins	0	0	NA	
	PCB-1248	0	0.0041	NA	
	PCB-1254	0	0.0092	NA	
	PCB-1260	0	0.0032	NA	

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.53 Screening of Risks to Soil Invertebrates from Sludge Application in Loblolly Pine Plantation

Exposure Scenario	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	2	2.25	60	0.0375
	Cadmium	0.2	0.743	20	0.0371
	Copper	10	24.8	60	0.413
	Lead	20	23.9	500	0.0478
	Mercury	0.05	0.136	0.1	1.36
	Molybdenum	1	1.23	NA	
	Nickel	10	11.6	200	0.0581
	Selenium	0.2	0.330	70	0.00471
	Zinc	20	69.6	200	0.348
	TCDD	0	8.58E-07	NA	
	PCB-1248	0	0.0032	NA	
	PCB-1254	0	0.0072	NA	
	PCB-1260	0	0.0025	NA	

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.54 Screening of Risks to Soil Invertebrates from Sludge Application in Deciduous Forest

Exposure Scenario	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	4.1	4.45	60	0.0741
	Cadmium	0.2	0.960	20	0.0480
	Copper	8.6	29.3	60	0.488
	Lead	48.5	54.0	500	0.108
	Mercury	0.051	0.171	0.1	1.71
	Molybdenum	3	3.32	NA	
	Nickel	15	17.3	200	0.0863
	Selenium	0.2	0.382	70	0.00545
	Zinc	61.5	131	200	0.655
	TCDD	0	1.20E-6	NA	
	PCB-1248	0	0.0045	NA	
	PCB-1254	0	0.0101	NA	
	PCB-1260	0	0.0035	NA	

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.55 Screening of Risks to Soil Invertebrates from Sludge Application in Rangeland

Exposure Scenario	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	2.2	2.45	60	0.0408
	Cadmium	0.55	1.09	20	0.0546
	Copper	4.84	19.6	60	0.327
	Lead	12.4	16.3	500	0.0326
	Mercury	0.054	0.14	0.1	1.4
	Molybdenum	0.1	0.331	NA	
	Nickel	11.5	13.1	200	0.0656
	Selenium	1	1.13	70	0.0161
	Zinc	39.84	89.5	200	0.447
	TCDD	0	8.58E-07	NA	
	PCB-1248	0	0.0032	NA	
	PCB-1254	0	0.0072	NA	
	PCB-1260	0	0.0025	NA	

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.56. Summary of Screening Results for Soil Invertebrates from Application of Sewage Sludge, 1 X 40 Mg/ha (Hazard Quotients above 1)

Ecosystem	NSSS 95% UCL
Douglas-fir forest	Hg[1]
Loblolly Pine Plantation	Hg[1]
Semiarid Rangeland	Hg[1]
Eastern Deciduous Forest	Hg[1]

[1] Chemical was eliminated as a COPEC in the risk characterization

The assumed background concentrations of mercury in the four ecosystems are: 0.051 mg/kg for the Douglas-fir, 0.051 mg/kg for the eastern deciduous forest, 0.050 mg/kg for the loblolly pine plantations, and 0.054 mg/kg for the semi-arid rangeland. If risks from background levels of mercury in soil were screened using the ORNL benchmark (0.1 mg/kg), the hazard quotient for all ecosystems would be about 0.5, suggesting no adverse effects. Background concentrations of mercury contribute significantly to the hazard quotients, but they are not the major contributor.

Thus, concentrations of mercury in all four sludge-treated ecosystems were above benchmarks for the soil invertebrate community. The toxicity of this element was further investigated in Sect. 4.4.3.1.1 to determine whether it was a credible contaminant of potential ecological concern (COPEC) for soil invertebrates in any ecosystem.

Effects of retained contaminants

Mercury. The ORNL benchmark for toxicity to soil invertebrates from mercury is based on two studies. Neither demonstrates effects at levels of mercury below the 0.14 to 0.17 mg/kg estimated concentrations in the assessment ecosystem soils. The 0.1 mg/kg benchmark is derived from a study of the survival and reproduction response of *Octochaetus pattoni* to the addition of mercuric chloride. Survival and cocoon production were reduced 65% and 40% when mercury was added to a level of 0.5 mg/kg in soil. Because the reduction in earthworm survival was 65%, a safety factor of 5 was used to obtain the benchmark of 0.1 mg/kg (Efroymson et al. 1997b). The other study that was described in Efroymson et al. (1997b) examined the effect of methyl mercury on the survival and segment regeneration of *E. foetida*. Both segment regeneration and survival were reduced (69% and 21%) at 12.5 mg/kg soil, and 2.5 mg/kg had no adverse effect on the worms. No distinction could be made between the toxicity of the methylated and chloride forms of the element.

Studies of the uptake of metals in sludge-treated soils by earthworms do not generally include mercury. Cadmium, copper, lead, zinc, and nickel are more commonly measured (Wade et al. 1982, Kruse and Barrett 1985, Beyer et al. 1982, Pietz et al. 1984, Hartenstein et al. 1980, Andersen 1979). Thus, the effect of sludge on the uptake of mercury by earthworms is not discussed.

Mercury was determined not to be a COPEC because: 1) hazard quotients for soil invertebrates in both ecosystems were less than 2 (i.e., close to no-hazard values), 2) toxicity to earthworms was never observed at the benchmark concentration, and 3) the background concentration of mercury is a substantial fraction of the total and is not likely to be as bioavailable mercuric chloride freshly added to soil.

Dioxin. Although there was no ORNL invertebrate toxicity benchmark for dioxin, the threshold for toxicity to *Allolobophora caliginosa* was observed elsewhere to be between 5 mg/kg and 10 mg/kg in soil (Reinecke and Nash 1984). Given that under the 95% UCL scenario, the assumed concentration of dioxin in soil ranged from 8.6×10^{-7} to 1.2×10^{-6} mg/kg, there are no risks from dioxin in sewage sludge at the 40 Mg/ha application rate. Indeed at the current concentrations of dioxin in sewage sludge, the safety factor is very large under the 40 Mg/ha application rate scenario. Dioxin was not retained as a COPEC.

4.4.3.2 Media toxicity tests

Ideally, the media toxicity test line of evidence would be a comparison of results from sludge-treated soils (applied at 40 Mg/ha) with those of untreated soils and (or perhaps fertilizer, if the ecosystem is ordinarily fertilized) from each of the four ecosystems. Instead, the available toxicity tests are of sludges without soil or of soils treated with metal-spiked sludge. The lowest concentrations of metals in sewage sludge without soil that may inhibit the growth of *E. foetida* are 480 mg/kg for mercury, 1100 mg/kg for nickel, 1200 mg/kg for copper, 1300 mg/kg for zinc, and 1800 mg/kg for cadmium (Hartenstein et al. 1981). The range of sludge-treated soil concentrations in the four ecosystems for mercury, nickel, copper, zinc, and cadmium in this assessment were assumed to be: 0.14 to 0.17 mg/kg, 12 to 17 mg/kg, 20 to 41 mg/kg, 70 to 180 mg/kg and 0.74 to 1.7 mg/kg, respectively. The range of toxic concentrations for mercury, 480-4800 mg/kg, is three to four orders of magnitude greater than the 0.1 mg/kg ORNL benchmark concentration in soil.

Thus, application of sewage sludge at 40 Mg/ha or to the above levels in soil does not pose a risk to soil invertebrates in any of the four ecosystems, based on the media toxicity test line of evidence alone.

4.4.3.3 Surveys of soil invertebrates

Two studies described in Sect. 4.4.2 apply dewatered sewage sludge at levels of 40 Mg/ha or less. In the study in Denmark (Andersen 1979), the numbers and biomass of various species of earthworms increased with sewage sludge addition, compared to control (potassium-treated) treatments. Sewage sludge from Aiken, SC increased the number of mites and springtails. In addition, at the Lafayette Experimental Station in New York State, where sludge was applied at 56 Mg/ha, earthworm density and/or biomass increased in a hardwood, Norway spruce, and old field ecosystem.

Based on this line of evidence alone, changes in the soil invertebrate community in the Douglas-fir forest, eastern deciduous forest, and loblolly pine plantations are likely with sludge addition. Earthworm populations are likely to increase. Litter decomposition may also increase. No surveys of the invertebrate community were undertaken in the semiarid rangeland.

4.4.4 Weight of Evidence

The weights of evidence for soil invertebrates in the Douglas-fir forest, loblolly pine plantation, and eastern deciduous forest are presented in Table 4.57. Based on the three lines of evidence, single chemical toxicity, media toxicity tests, and surveys of invertebrates, no distinction in effects on assessment endpoints (reduction in abundance or growth of soil invertebrates, change in relative abundance of taxonomic groups) can be made among the three forest ecosystems. Sewage sludge has the potential to alter the soil invertebrate community in all three ecosystems, primarily as a result of nutrients in sewage sludge. The weight of evidence at the 40 Mg/ha application rate concludes that there is an effect on the relative abundance of different taxonomic groups; the effect is a risk if it is deemed that any community shift constitutes a risk. However, there is not enough evidence to evaluate impacts on the invertebrate role in litter decomposition or the duration of the community change. Metal and organic contaminants in sewage sludge are probably not responsible for the soil invertebrate community changes.

Table 4.57. Weight of evidence for soil invertebrate communities in the Douglas-fir forest, loblolly pine plantation, and deciduous forest, 1 X 40 Mg/ha application

Endpoint	Evidence	Result	Risk or Effect[1]	Explanation
Reduction in abundance or growth of soil invertebrates	Single Chemical Toxicity Data	–		Concentrations of chemicals in sewage sludge-treated soils do not exceed toxicity benchmarks, except for mercury in all ecosystems. Based on reasons stated in the risk characterization, mercury is not a Contaminant of Ecological Concern.
	Media toxicity tests	–		A toxicity test evaluating earthworm growth in sludge resulted in toxicity thresholds that were much higher than the concentrations of elements in sludge-treated soil at the application rate assumed in this assessment.
	Surveys of biota	–		De-watered sludge applied to a loblolly pine plantation in South Carolina; a mixed hardwood, Norway spruce, and old field in New York state; a red pine plantation in Wisconsin; and a field in Denmark increased earthworm density and/or biomass.
	Weight of evidence	–	Risk	Reduction in abundance or growth of soil invertebrates is unlikely.
Change in relative abundance of taxonomic groups of soil invertebrates	Surveys of Biota	+		Changes in the soil invertebrate community (e.g., increases in particular earthworm species, increases or decreases in mite populations, increases in springtail populations, increases in nematodes) have been observed in a few sludge-treated ecosystems. Whether these constitute risks depends on the functions of the affected organisms and the management goals for the forest. The duration of the change is unknown. The shift is probably a result of nutrients rather than the metal or organic contaminants in sewage sludge.
	Media toxicity tests	+		In soil cores, the abundance of one species of earthworm was dependent on the presence of another. Nematode populations decreased in the presence of sewage sludge.
	Weight of evidence		Effect	

[1] *Risk* indicates an adverse effect; *effect* indicates a change in the community that may or may not be considered adverse.

Although the 95% UCL values of the contaminant concentrations in the NSSS were screened for risks to soil invertebrates in the semi-arid rangeland, the ORNL benchmarks are not appropriate for the evaluation of risks to ants, termites, and other soil invertebrates in this ecosystem. Since no surveys or toxicity tests were conducted using western soils or common arthropods from these regions, a weight of evidence for risks to soil invertebrates in the semi-arid rangeland is not presented.

4.4.5. Uncertainties

- **Background concentrations.** Background concentrations of chemicals in soil were often responsible for a substantial fraction of the estimated toxicity in sludge-treated ecosystems. Background levels varied significantly among the ecosystems in this study. The spatial variability of background levels within an ecosystem and that within sludge-treated ecosystems in the United States have not been incorporated in this assessment. The likelihood of mobilization of background levels of chemicals by the application of sewage sludge is unknown.

- **Incorporation of sewage sludge.** The rate of incorporation of chemicals in sewage sludge into soils is unknown, as is the conservatism of the assumption that the sewage sludge is mixed in the surface 15 cm of soil.

- **Bioavailability of Contaminants.** The potential difference in bioavailability between chemicals in sludge-treated soil and chemicals in untreated soil has been discussed previously. It is also notable that bioavailability may vary from sludge to sludge and from soil to soil. Toxicity tests rarely came from the particular assessment ecosystems for which they were used.

- **Decomposition of sludge and bioavailability of chemicals.** The bioavailability of metals and other chemicals applied to soils in sewage sludge may be slowly altered with the decomposition of organic matter in sludge.

- **Extrapolation of ambient media toxicity test data.** It has been assumed that media toxicity tests and biota surveys at application rates of 40 Mg/ha are comparable to the assumptions in this assessment. To determine if this is true, the concentrations of active chemicals (particular nutrients or contaminants) in sludge would have to be measured, as well as the ultimate concentrations in soil.

- **Extrapolation from Published Soil Invertebrate Toxicity Data.** Published toxicity studies emphasize earthworms rather than other invertebrate species . Toxicity thresholds for non-earthworm invertebrates n in assessment ecosystems are likely to vary from those in studies of earthworms.

- **Single Contaminant Tests vs Exposure to Multiple Contaminants in the Field.** While organisms in the four ecosystems are exposed to multiple contaminants concurrently, published toxicological values from which ORNL benchmarks were derived only consider effects experienced from exposures to single contaminants. Because some contaminants to which invertebrates are exposed can interact antagonistically, single contaminant studies may

overestimate their toxic potential. Similarly, for those contaminants that interact additively or synergistically, single contaminant studies may underestimate their toxic potential.

- **Inorganic Forms or Species Present in the Environment.** Toxicity of metal species varies dramatically depending upon the valence state or form (organic or inorganic) of the metal. Relative toxicity to soil invertebrates are generally unknown. The NSSS does not report species or form of contaminants in municipal sewage sludges.

4.5 RISKS TO MICROBIAL PROCESSES AND MICROORGANISMS

Three assessment endpoints related to microorganisms and microbial processes are: 1) microbial biomass, 2) decomposition of native soil organic matter and plant litter, and 3) nitrogen cycle processes. All changes in measurement endpoints for these entities are reported in the weight of evidence.

4.5.1 Exposure Assessment for Microbial Processes

The general exposure assumptions for the ecological risk assessment are stated in Section 4.1. A critical assumption for soil microbial processes is that constituents of sludge are evenly distributed through the top 15 cm of soil. For the purposes of this study, it is assumed that if microorganisms reside in underlying soils or in the litter layer, their exposures are similar to those in the surface 15 cm of soil. The potentially different exposures in the plant rhizosphere are also not considered. The soil concentrations of contaminants to which the microorganisms are exposed are presented in Tables 4.2, Douglas-fir forest; 4.3, loblolly pine plantations; 4.4, deciduous forest; and 4.5, semi-arid rangeland.

Various species of microorganisms have been observed to alter their microenvironments, thus changing the bioavailability of metals in soil. For example, fast-growing rhizobia usually decrease the pH of their environment, enhancing metal availability and toxicity (Angle et al. 1993). On the other hand, bradyrhizobia may cause an alkaline reaction and excrete a thick polysaccharide capsule, both actions reducing the bioavailability of metals (Angle et al. 1993).

4.5.2 Effects Assessment for Microbial Processes

4.5.2.1 Single chemical soil toxicity

ORNL benchmarks

Oak Ridge National Laboratory (ORNL) has developed benchmarks for screening contaminant concentrations in soils for potential risks to soil microbial processes (Efroymson et al. 1997b). The benchmarks were derived from studies of carbon mineralization, nitrogen transformation, and enzyme activities. Most data were from toxicity studies in the laboratory with soil or soil and litter, but not in culture media. Exposure durations were from one and one-half hours to one and one-half years. The chemicals were often added in the form of salts, and sewage sludge was never the source. The bioavailability of contaminants in sewage sludge is discussed in Section 4.1.4, and the difference between the bioavailability of soluble salts and contaminants in sewage sludge is probably not equivalent for microorganisms and plants.

The method used for deriving benchmarks for toxicity to microbial processes (Efroymson et al. 1997b) is based on the National Oceanographic and Atmospheric Administration's method for deriving the Effects Range Low (ER-L) (Long and Morgan, 1991), which has been recommended as a sediment screening benchmark by U.S. Environmental Protection Agency (EPA) Region IV. The ER-L is the tenth percentile of the distribution of various toxic effects thresholds for various organisms in sediments. (A threshold is defined as a 20% effects level (see Sect. 1.5.1).) Thus, the assessor using the benchmarks is 90% certain that microbial processes in the site soil are protected. As with NOAA ER-Ls, statistical fitting was not used in the derivation of benchmarks because there were seldom sufficient data (Efroymson et al. 1997b).

The level of confidence in the benchmarks ranges from low (e.g., arsenic benchmark derived from 8 values in 2 studies) to high (e.g., zinc value derived from 47 values in 14 studies) (Efroymson et al. 1997b). All of the benchmarks are conservative.

Benchmarks were used first to screen for Contaminants of Potential Ecological Concern, i.e., to eliminate contaminants from further consideration for risks to microbial processes if they did not exceed benchmark concentrations in soils) and to provide a conservative (if somewhat poor) estimate of risks from particular constituents of sewage sludge. Both the screening of contaminants for risks and the use of benchmarks for toxicity to microorganisms in the weight of evidence are presented in the risk characterization for each ecosystem (Sect. 4.5.3).

4.5.2.2 Media toxicity tests

Both toxicity to microorganisms and effects on microbial processes (which could be positive or negative) such as nitrification and decomposition are the subject of these studies. Experiments are not described here if they measured nitrogen or carbon transformations of sewage sludge itself during short time periods.

Synthetic (digestion of *Candida utilis* cells) sludge was added to soil at 22.4 Mg/ha (Terry et al. 1979). The decomposition of native soil organic matter was increased by 100% at 336 days post-application.

Sewage sludge was mixed with sandy or clayey soils in pots to concentrations of 0, 15, and 30 g of sludge per kg of soil (Lindemann and Cardenas 1984). (In the present study, the mixture was 19 to 24 g of sludge/kg). The total mineralization of nitrogen increased with increasing application rate, but the magnitude of the increase in denitrification was even larger; thus net mineralization decreased with increased sludge application rate. Total and net mineralization were greater in the clayey than in the sandy soil.

Sewage sludge and ammonium nitrogen (100 mg/kg) were mixed with Cecil sandy loam at rates of 0, 1, 4, or 16 g/kg (Wilson 1977). (In the present study, the mixture was 19 to 24 g of sludge/kg). Samples were taken every week for six weeks and analyzed for ammonium nitrogen and nitrate/nitrite nitrogen. At the highest rate, nitrification was reduced at least 20% compared to the non-amended control. However, because the experiment ended at six weeks, it is unclear whether the difference would have persisted.

Sewage sludge (100 g) was mixed with 1 kg of soil in plant pots containing either a fine-textured volcanic soil or a terra rossa, rich in iron oxides and kaolinite clay, both from Italy (Coppola et al. 1988). Cadmium sulfate was added to the sludge to obtain a level of 0, 2, 4, 8, or 16 mg Cd/kg of soil. (In the present study, the cadmium concentration in soil ranges from 0.7 to 1.7 mg/kg.) Soil microbial activities were measured after growth of ryegrass in the pots. Soil biomass, as measured by ATP, and mineralization of organic carbon were not generally different among treatments, except that the former was lowered in the volcanic soil at 16 mg/kg cadmium. Of the counts of microbial functional groups taken (ammonifiers, ammonium-oxidizers, nitrite-oxidizers, free-living N_2 fixers, aerobic microorganisms and anaerobic microorganisms), the ammonifiers were the most sensitive to Cd, with counts decreasing with Cd addition, especially in volcanic soil. Of the nitrifiers, only nitrite-oxidizers were reduced by Cd in the volcanic soil. However, nitrification was not inhibited in any samples. Nonsymbiotic soil nitrogen fixation was substantially reduced with increased cadmium concentration. Of the effects noted above, only values for ATP and nitrogen fixation were measured at less than 80% of control values at the lowest cadmium level tested, 2 mg/kg. However, the ATP increased to control levels at 4 mg/kg Cd, and the lowered nitrogen fixation at 2 mg/kg was only observed in the terra rossa soil.

Sewage sludge was added at 200 Mg/ha to soils at the Gleadthorpe Experimental Husbandry Farm in England (Chander and Brookes 1993). Soil biomass carbon, measured by the fumigation-extraction method, was at least 20% less than control soils when sludge was supplemented with zinc, giving a soil concentration of 705 mg/kg; when it was supplemented with copper, giving a soil concentration of 415 mg/kg; and when it was supplemented with zinc and copper, to soil concentrations of 367 and 191 mg/kg, respectively.

Soils from the Market Garden Experiment at the Woburn Experimental Farm, to which sewage sludge was added from 1942-1961, and low-metal soils that received manure from 1947 to 1967, were evaluated for the ability to support cyanobacteria (Brookes et al. 1986). High-metal, sludge-treated soils, containing 289 mg/kg zinc, 102 mg/kg copper, 27 mg/kg nickel, and 8.6 mg/kg cadmium, was associated with poor acetylene reduction (much longer lag and 1/3 of rate of control), indicative of nitrogen fixation. The low metal soil (control) contained 92 mg/kg zinc, 27 mg/kg copper, 4 mg/kg nickel, and 2.2 mg/kg cadmium. In addition, the low-metal soil fixed about 10 times as much ^{15}N-labeled N_2 as the high-metal soil.

Soils from the Market Garden Experiment at the Woburn Experimental Farm, to which sewage sludge was added from 1942-1961 at 16.4 Mg/ha/yr, and low-metal soils that received manure from 1947 to 1967, were evaluated for ability to support vesicular-arbuscular mycorrhizae (Koomen et al. 1990). White clover was planted in each pot. The percent infection of farmyard manure plots versus sludge-treated pots by native mycorrhizae was 60% versus 1%. However, there was no significant difference in shoot and root weight of clover in the two types of pots. When an inoculum of the species *Glosmus mosseae* was added to the pots, there was no infection in the sludge-treated pots, compared to 46% in the farmyard manure pots. The sludge-treated soil contained 381 mg Zn/kg, 126 mg Cu/kg, 42 mg Ni/kg, 11.1 mg Cd/kg, 109 mg available P/kg, and 1.99 % C. The manure-treated soil contained 94 mg Zn/kg, 24 mg Cu/kg, 16 mg Ni/kg, 1.6 mg Cd/kg, 112 mg available P/kg, and 1.42 % C.

Soils from the Market Garden Experiment at the Woburn Experimental Farm described above were evaluated for the growth and N_2 fixation of white clover (*Trifolium repens*) (McGrath et al. 1988). Root nodules on plants grown on sludge-treated soils were small, white,

numerous, and ineffective in N_2-fixation. On the other hand nodules on plants in manure-treated soils were pink (because of leghemoglobin), large, few in number, and effective. The two soils were mixed, and a 50% reduction in nitrogen fixation was obtained from a soil containing: 334 mg Zn/kg, 99 mg Cu/kg, 27 mg Ni/kg, and 10 mg Cd/kg of soil. These results were confirmed by Angle et al. (1993), who note that *Rhizobium leguminosarum* bv. *trifolii* in old sludge-treated soils are ineffective. Hirsch et al. (1993) found that the ineffective rhizobium was a single strain, ineffective on white and red clover and unable to nodulate *Vicia hirsuta*, but effective on subterranean clover. Thus, the sludge-treated soils reduced the genetic diversity of *Rhizobium leguminosarum* bv. *trifolii* populations.

Plots in Ultuna, Uppsala, Sweden were treated every second year with sewage sludge at a rate equivalent to 4 Mg/ha ash-free organic matter from 1956 to 1988 (Martensson and Witter 1990). (The total application rate was probably more than 8 Mg/ha, giving an application rate of at least 128 Mg/ha.) Sludge-treated plots had lower numbers of *Rhizobium leguminosarum* biovar*trifolii* than unfertilized control plots but not than ammonium sulfate treated plots. The authors suggested that the low pH (<6) was responsible for the low numbers of rhizobia. Although the sludge-treated site showed a delay in nodulation, the number of nodules, dry weight of plants or nitrogen content of plants were unaffected. In sludge-treated soils under permanent light in Petri dishes, blue-green algae appeared after 3 months, compared to two weeks for the untreated soil. Maximum acetylene-reductase activity was about 100 times lower in this treatment than in others. Cyanobacteria were never observed in the ammonium sulfate-fertilized soil. No significant differences in free-living nitrogen fixation activity were observed between any treatments.

It is often suggested that microbial populations adapt to high metal concentrations. For example, the tolerance of isolates of *Rhizobium leguminosarum* biovar *trifolii* from nodules of clover plants grown on sludge-treated soils at Woburn Experimental Farm, Rothamsted, England, to copper, nickel, cadmium, and zinc was greater than for isolates from manure-treated plots (Chaudri et al. 1992). Barkay et al. (1985) noted that gram-positive bacteria were not present in cadmium-resistant communities from sludge-treated soils.

4.5.2.3 Miscellaneous surveys of microbial processes

Few surveys of microbial processes in sludge-treated plots have been undertaken. Most effects to microorganisms are determined in pot studies and are described in Sect. 4.5.2.2. Exceptions are the studies below and those rangeland studies described in Sections 2.6.1 (Rio Puerco Valley), 2.6.2 (Sevilleta National Wildlife Refuge), and 2.6.6 (Utah).

Maryland hardwood forest

Aerobically-digested municipal sewage sludge was applied to a hardwood forest in Maryland at rates of 0, 3, 6, and 12 Mg/ha, corresponding to nitrogen loading rates of 0, 200, 400, and 800 Mg N/ha, respectively (Aschmann et al. 1990). Nitrogen in the standing litter of the 12 Mg/ha plot increased the first year following application, though foliar nitrogen content did not. The authors use these results to suggest that decomposition rates increased at the highest application rate, or sludge-borne nitrogen was sorbed to the litter.

Reclaimed stripmines

Five mine sites reclaimed with 120 to 134 Mg/ha sewage sludge were 1 to 5 years old at the time of this study (Seaker and Sopper 1988). The soils were planted with grass or legumes. Numbers of aerobic bacteria, fungi, *Nitrobacter*, *Nitrosomonas*, and actinomycetes were reported, as well as decomposition rates. Values for a control site or for microbial measurements prior to application were not presented. Bacterial and fungal populations and soil community respiration were higher on sludge-amended sites than on fertilizer-amended plots, with the 1-year-old site having the highest population of each. Actinomycete populations were higher on the older sites than on the 1-year-old plot, and for older plots their numbers were higher than in fertilizer-amended plots. *Nitrosomonas* and *Nitrobacter* populations were several orders of magnitude higher on the sludge-treated plots than on fertilized plots. The former genus did not vary among different-aged plots, but the latter had higher numbers on the youngest site. On sludge-amended plots, decomposition rate increased with the age of the plot. For example, only half of a grass sample was decomposed on the one-year-old plot, compared to 97% on the 5-year-old site. This observation contrasts with what might be expected from the microbial population results.

German field soil and woodland soil

High- or low-metal-containing sewage sludge was added to an arable and old woodland soil from Braunschweig, Germany, at rates of 5 and 15 Mg/ha/yr for 10 years (Fliessbach et al. 1994). In general, soil microbial biomass carbon increased with sludge application, though at the higher application rate, sludge containing the higher metals resulted in lower biomass than the lower metal sludge. Soil respiration in the high-application-rate plots also increased.

4.5.3 Risk Characterization for Microbial Processes

As stated in Sect. 4.2.3, in the Risk Characterization, the results of the exposure assessment (Sect. 4.5.1) and effects assessment (Sect. 4.5.2) are integrated to estimate risks based on each line of evidence, and then a weight of evidence inference logic is applied to each assessment endpoint. Although all three types of evidence were available for this assessment, the types of studies were not ideal. Single chemical toxicity tests from the literature (i.e., ORNL benchmarks) did not account for the effect of sewage sludge on bioavailability. Surveys of microorganisms and microbial processes were often conducted at application rates substantially different from the 40 Mg/ha assumed here, and changes occurred in the short term that may not be extrapolated to the long term. The long term experiment from Rothamsted, England is useful, however. Media toxicity tests were undertaken, sometimes with soluble salts added.

4.5.3.1 Single chemical soil toxicity data

Estimated concentrations of sewage sludge-amended soil in the Douglas-fir forest, loblolly pine plantation, eastern deciduous forest, and semi-arid rangeland are compared to toxicity benchmarks in Tables 4.58, 4.59, 4.60, and 4.61, respectively. According to the screen against ORNL toxicity benchmarks, potential risks to microbial processes in the Douglas-fir forest and the eastern deciduous forest may exist from exposure to zinc under the NSSS-based exposure scenario at an application rate of 40 Mg/ha (Table 4.62). The hazard quotients for zinc are: 1.8 for the Douglas-fir forest (Table 4.58) and 1.3 for the deciduous forest (Table 4.60).

The hazard quotient was less than 2 in both cases, and in the eastern deciduous forest, it should be rounded down to 1, given the single significant digit of the benchmark.

As stated in Sect. 4.3.3.1, the assumed background concentrations of zinc in the four ecosystems are: 120 mg/kg for the Douglas-fir, 61.5 mg/kg for the eastern deciduous forest, 20 for the loblolly pine plantations, and 40.7 for the semi-arid rangeland. If risks from background levels of zinc in soil were screened using the ORNL benchmark for microbial processes, potential risks would be predicted for the Douglas-fir forest (hazard quotient = 1.2). Background concentrations of zinc contribute significantly to all hazard quotients, and whether or not there are risks to microorganisms and/or microbial processes depends in large part on whether the background metals are mobilized in the long term. The toxicity of zinc was further investigated in Sect. 4.5.3.1.1 to determine whether it was a credible contaminant of potential ecological concern (COPEC) in either of the two ecosystems.

4.5.3.1.1 Effects of Retained Contaminants

Zinc

The ORNL benchmark of 100 mg/kg zinc in soil is the 10th percentile of 47 reported thresholds for toxicity of zinc (Efroymson et al. 1997b). Among these thresholds, several were below the assessment concentrations of zinc in the Douglas-fir and eastern deciduous forest soils. Premi and Cornfield (1969, 1969/1970) added zinc sulfate to a sandy loam soil and observed reduced nitrification at 100 mg/kg. Zinc chloride was added to sand, sandy loam, silty loam, clay, and sandy peat soils (Doelman and Haanstra 1986). Soil urease activity was reduced 50% at 110, 90, and 70 mg/kg for the sandy loam, clay, and sandy peat soils eighteen months after addition of the zinc. Effective concentrations were much higher after only 6 weeks. Phosphatase activity was 50% inhibited in sand at 170 mg/kg (Doelman and Haanstra 1989)

The soil at Pack Forest described by Greenleaf-Jenkins and Zasoski (1986) is an Everett gravelly sandy loam. Thus the soil texture is not very different from the soils used in the studies above.

McGrath et al. (1995) reviewed adverse effects of sewage sludge on microbial activity. They described effects on N_2-fixation in cyanobacteria at zinc concentrations ranging from 26 to 305 mg/kg soil. Other metal contaminants such as copper are also present at high concentrations, and the most toxic metals were not determined. The population of *Rhizobium leguminosarum* biovar *trifolii* was reduced at zinc levels ranging from 130 to 180 mg/kg soil. Soil microbial biomass was reduced at zinc concentrations ranging from 180 to 857 mg/kg soil.

There is not sufficient evidence to eliminate zinc as a COPEC for the Douglas-fir or eastern deciduous forest. Therefore, reductions in soil microbial processes, including nitrification, soil urease activity, and/or phosphatase activity, may result from exposure to zinc. Based on the single chemical soil toxicity line of evidence alone, sludge is not expected to adversely impact the microbial community in the semi-arid rangeland or the loblolly pine plantations.

Table 4.58 Screening of Risks to Microbial Processes from Sludge Application in Douglas-fir forest

Exposure Scenario	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	2.6	2.92	100	0.0292
	Cadmium	1	1.69	20	0.0845
	Copper	22	40.8	100	0.408
	Lead	45	50.0	900	0.0556
	Mercury	0.051	0.160	30	0.00534
	Molybdenum	3	3.29	200	0.0165
	Nickel	14	16.1	90	0.178
	Selenium	0.2	0.365	100	0.00365
	Zinc	120	183	100	1.83
	chlorinated dioxins	0	0	NA	
	PCB-1248	0	0.0041	NA	
	PCB-1254	0	0.0092	NA	
	PCB-1260	0	0.0032	NA	

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.59 Screening of Risks to Microbial Processes from Sludge Application in loblolly pine plantation
Exposure Scenario

	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	2	2.25	100	0.0225
	Cadmium	0.2	0.743	20	0.0371
	Copper	10	24.8	100	0.248
	Lead	20	23.9	900	0.0266
	Mercury	0.05	0.136	30	0.00453
	Molybdenum	1	1.23	200	0.00616
	Nickel	10	11.6	90	0.129
	Selenium	0.2	0.330	100	0.00330
	Zinc	20	69.6	100	0.696
	TCDD	0	8.58E-7	NA	
	PCB-1248	0	0.0032	NA	
	PCB-1254	0	0.0072	NA	
	PCB-1260	0	0.0025	NA	

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.60 Screening of Risks to Microbial Processes from Sludge Application in eastern deciduous forest

Exposure Scenario

	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	4.1	4.448	100	0.0445
	Cadmium	0.2	0.960	20	0.0480
	Copper	8.6	29.3	100	0.293
	Lead	48.5	54.0	900	0.0600
	Mercury	0.051	0.171	30	0.00571
	Molybdenum	3	3.32	200	0.0166
	Nickel	15	17.3	90	0.192
	Selenium	0.2	0.382	100	0.00382
	Zinc	61.5	131	100	1.31
	TCDD	0	1.20E-6	NA	
	PCB-1248	0	0.0045	NA	
	PCB-1254	0	0.0101	NA	
	PCB-1260	0	0.0035	NA	

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.61 Screening of Risks to Microbial Processes from Sludge Application in semiarid rangeland
Exposure Scenario

	Contaminant	Background (mg/kg)	Post-sludge Conc[1] (mg/kg)	Plant Benchmark	Hazard Quotient
NSSS 95% UCL	Arsenic	2.2	2.45	100	0.0245
	Cadmium	0.55	1.09	20	0.0546
	Copper	4.84	19.6	100	0.196
	Lead	12.4	16.3	900	0.0181
	Mercury	0.054	0.14	30	0.00467
	Molybdenum	0.1	0.331	200	0.00166
	Nickel	11.5	13.1	90	0.146
	Selenium	1	1.13	100	0.0113
	Zinc	39.84	89.5	100	0.895
	TCDD	0	8.58E-07		
	PCB-1248	0	0.0032		
	PCB-1254	0	0.0072		
	PCB-1260	0	0.0025		

[1]Soil concentration based on estimated chemical concentration in top 15 cm of soil, following 1x application of 40 Mg/ha

Table 4.62. Summary of Screening Results for Microbial Processes from Application of Sewage Sludge, 1 X 40 Mg/ha (Hazard Quotients above 1)

Ecosystem	NSSS 95% UCL
Douglas-fir forest	Zn
Loblolly Pine Plantation	none
Semiarid Rangeland	none
Eastern Deciduous Forest	Zn

4.5.3.2 Media toxicity tests

Ideally, the media toxicity test line of evidence would be a comparison of effects of sludge treatment at 40 Mg/ha on multiple microbial processes with results from untreated soils (or perhaps fertilized soils, if the ecosystem is ordinarily fertilized) from each of the four ecosystems. Instead, a series of tests have been performed using soils from various application sites, some with soluble salts added, usually at much higher rates of application than 40 Mg/ha. The most extensive work has been done at the Woburn Experiment Farm in England, where sewage sludge was applied from 1942 to 1961; thus, long-term effects of sludge addition have been studied at that site. However, the cumulative loading, 324 Mg/ha, was much higher than the 40 Mg/ha rate assumed in this assessment. Thus, results from that study are not directly applicable to the risk assessment.

The studies that are relevant to the application rate of the assessment are the following. Decomposition of native organic matter was increased by the application of synthetic sewage sludge at 22.4 Mg/ha (Terry et al. 1979). This study differed from the many experiments that investigated decomposition of sludge itself. In another study, both the mineralization of nitrogen and denitrification were accelerated with the addition of sewage sludge. Because the latter process was enhanced to a greater degree, net mineralization decreased with increased sludge application rate.

In a sewage sludge to which 2 mg/kg of cadmium was added, the rate of nonsymbiotic nitrogen fixation was reduced 46% over control values in one of two soils (Coppola et al. 1988). In this risk assessment, the assumed levels of cadmium in soil range from 0.74 mg/kg to 1.7 mg/kg, the latter value for the Douglas-fir forest. Because this is a single study, and the soil is substantially different from that in the Douglas-forest, the effect of cadmium on nitrogen fixation in the Douglas-fir forest at 1.7 mg/kg is unknown.

Few conclusions can be made about effects of sewage sludge on microbial processes, using the toxicity test line of evidence alone and the assumed application rate. The only effect described above that is likely to be ecosystem-independent is the increased decomposition of the native organic matter in soil. The balance between nitrification and denitrification described above may vary from ecosystem to ecosystem. The 46% decrease in nonsymbiotic nitrogen fixation at 2 mg Cd/kg of soil could be extrapolated to the 1.7 mg/kg Cd level in the Douglas-fir forest, but given the differences in soil type, this extrapolation would require more evidence.

4.5.3.3 Surveys of microorganisms and microbial processes

The surveys that are most relevant to impacts from sewage sludge applied at 40 Mg/ha are those from the semi-arid rangeland. Impacts of sewage sludge amendment on microbial processes and biomass in the rangeland have been investigated rather extensively (Sect. 2.2). In Rio Puerco Valley microbial populations including ammonium oxidizers, other bacteria, and fungi increased linearly with increasing sewage sludge amendments (Dennis and Fresquez 1989); however, the diversity of fungi decreased in the short term (Fresquez and Dennis 1990). Decomposition of the litter of one plant accelerated, whereas that of another plant decreased with sludge application; the difference may have been attributable to the C:N ratios of the litter or to the relative fractions of water-soluble constituents in the plants (Aguilar et al. 1994b, Loftin and Aguilar 1994).

The colonization of sagebrush by mycorrhizae was reduced at 60 Mg/ha applications but increased at 30 Mg/ha applications of sewage sludge in Utah (Sabey et al. 1990). These are vesicular-arbuscular mycorrhizal (VAM) fungi. VAM colonization may be inhibited by heavy metals or by high soil or plant concentrations of nitrogen or phosphorus (Gerry O'Neill, Oak Ridge National Laboratory, personal communication, August 1996). As stated in Sect. 1.3.3.3, VAM mycorrhizae are important to species such as maples located on low nutrient soils. In the eastern deciduous forest they also form associations with dogwoods. All conifers, oaks, hickory and beech are colonized by ectomycorrhizal fungi (Basidiomycetes and Ascomycetes). These fungi are more often inhibited by nitrogen and phosphorus than by heavy metals.

With sewage sludge added at the rate of 5 Mg/ha/yr for 10 years, soil microbial biomass increased in soils from Germany (Fliessbach et al. 1994)

Thus, based on the biota survey line of evidence alone, sludge application at 40 Mg/ha in the Douglas-fir forest, eastern deciduous forest, and semi-arid rangeland is expected to 1) increase microbial biomass, at least in the short term, and 2) to selectively increase the decomposition of litter. VAM mycorrhizae may be affected in the rangeland and in the eastern deciduous forest, but the threshold for effects is probably close to the application rate assumed in this study and dependent on specific metal and nutrient concentrations in sewage sludge.

In the semi-arid rangeland, sludge application will 1) increase ammonium oxidizers, other bacteria, and fungi, at least in the short term, and 2) selectively increase the decomposition of litter. Effects on mycorrhizae at 40 Mg/ha are not predictable based on the differing results associated with 30 and 60 Mg/ha application rates.

4.5.4 Weight of Evidence

A weight of evidence is presented for several assessment endpoints for all ecosystems for which there is sufficient evidence: 1) change in microbial biomass, 2) change in decomposition (rate or equilibrium extent) of plant litter, 3) change in balance of processes in nitrogen cycle, 4) change in microbial diversity, and change in colonization of trees or shrubs by vesicular-arbuscular mycorrhizal fungi. The nitrogen cycle processes (e.g., nitrification) are not distinguished because of a lack of data at the application rate considered in this assessment.

Impacts from sewage sludge on all of the assessment endpoints are designated as "effects" rather than risks, based on a lack of regulatory and site-specific management guidance concerning what is to be protected.

4.5.4.1 Douglas-fir forest

The weight of evidence for the Douglas-fir forest is presented in Table 4.63. Based on evidence from surveys of soil microorganisms and processes in Germany and New Mexico, the microbial biomass is expected to increase with the addition of sewage sludge. This effect is intuitive, given the nutrient input from sewage sludge. What is not known is the duration of the effect on particular components of the microbial community.

Based on evidence from media toxicity tests and a survey of litter decomposition in the semiarid rangeland, the decomposition of some types of plant litter may accelerate with sludge

application. The decomposition rate of native soil organic matter may also be affected, but no test of municipal sewage-sludge-amended media was available.

Based on the ORNL benchmark for the toxicity of zinc to microbial processes, including nitrification, the addition of sewage sludge may impact the nitrogen cycle in the Douglas-fir forest. Toxicity tests at higher application rates than that which was assumed in this assessment have shown that nitrogen fixation, nitrification, and denitrification rates can be altered with sewage sludge amendment. Based on the weight of evidence, sewage sludge added at 40 Mg/ha is likely to impact the nitrogen cycle in the Douglas-fir forest.

Table 4.63. Weight of evidence for soil microorganisms in the Douglas-fir forest, 1 X 40 Mg/ha application

Endpoint	Evidence	Result	Risk or Effect[1]	Explanation
Change in microbial biomass	Biota Surveys	+	Effect	Microbial biomass increases with the addition of sewage sludge. The duration of the effect is unknown.
Change in decomposition (rate or equilibrium extent) of plant litter	Survey of microbial processes	±	Effect	A survey of litter decomposition in New Mexico suggests that litter decomposition may increase in the presence of sewage sludge.
Change in balance of processes in nitrogen cycle	Single Chemical Toxicity (Benchmarks)	+		Based on the ORNL benchmark for toxicity of zinc to microbial processes, including nitrification, the addition of sewage sludge may impact the nitrogen cycle.
	Media toxicity tests	±		Based on toxicity tests at higher application rates of sewage sludge, rates of nitrogen fixation, nitrification, and denitrification may be altered.
	Weight of Evidence	+	Effect	Sewage sludge added at 40 Mg/ha is likely to impact the nitrogen cycle in the Douglas-fir forest.

[1] *Risk* indicates an adverse effect; *effect* indicates a change in the community that may or may not be considered adverse.

4.5.4.2 Loblolly pine plantation

The weight of evidence for the loblolly pine plantation is presented in Table 4.64. As in the Douglas-fir forest, based on evidence from soil microorganism surveys in Germany and New

Table 4.64. Weight of evidence for soil microorganisms in the loblolly pine plantation, 1 X 40 Mg/ha application

Endpoint	Evidence	Result	Risk or Effect[1]	Explanation
Change in microbial biomass	Biota Surveys	+	Effect	Microbial biomass increases with the addition of sewage sludge. The duration of the effect is unknown.
Change in decomposition (rate or equilibrium extent) of plant litter	Survey of Microbial processes	±	Effect	A survey of litter decomposition in New Mexico suggests that litter decomposition may increase in the presence of sewage sludge.
Change in balance of processes in nitrogen cycle	Single Chemical Toxicity (Benchmarks)	–		Based on the ORNL benchmarks for toxicity of chemicals to microbial processes, the addition of sewage sludge is unlikely to impact the nitrogen cycle.
	Media toxicity tests	±		Based on toxicity tests at higher application rates of sewage sludge, rates of nitrogen fixation, nitrification, and denitrification may be altered.
	Weight of Evidence	0	Effect	The impact of sewage sludge added at 40 Mg/ha to the nitrogen cycle in the loblolly pine plantation is unknown.

[1] *Risk* indicates an adverse effect; *effect* indicates a change in the community that may or may not be considered adverse.

Mexico, the microbial biomass increases with the addition of sewage sludge. This effect is intuitive, given the nutrient input from sewage sludge. What is not known is the duration of the effect of particular components of the microbial community.

Based on evidence from media toxicity tests and vegetation surveys, the decomposition of some types of plant litter may accelerate with sludge application. The decomposition rate of native soil organic matter may also be affected, but no test of municipal sewage-sludge-amended media was available.

Based on the ORNL benchmarks for the toxicity of chemicals to microbial processes, non-nutrient contaminants in sewage sludge are unlikely to impact the nitrogen cycle in the Douglas-fir forest. Toxicity tests at higher application rates than that which was assumed in this assessment have shown that nitrogen fixation, nitrification, and denitrification rates can be altered with sewage sludge amendment. The impact of sewage sludge added at lower rates is unknown.

4.5.4.3 Eastern deciduous forest

The weight of evidence for the deciduous forest is presented in Table 4.65. Based on evidence from soil microorganism surveys in Germany and New Mexico, the microbial biomass increases with the addition of sewage sludge. This effect is intuitive, given the nutrient input from sewage sludge. What is not known is the duration of the effect of particular components of the microbial community. Some evidence suggests that the colonization of vesicular-arbuscular mycorrhizal fungi may be reduced with application of sewage sludge.

Table 4.65. Weight of evidence for soil microorganisms in the eastern deciduous forest, 1 X 40 Mg/ha application

Endpoint	Evidence	Result	Risk or effect[1]	Explanation
Change in microbial biomass	Biota Surveys	+	Effect	Microbial biomass increases with the addition of sewage sludge. The duration of the effect is unknown.
Change in extent of colonization of trees by vesicular-arbuscular mycorrhizal fungi	Biota Surveys	±	Effect	Colonization of sagebrush by VAM fungi is reduced, with the threshold application rate between 30 Mg/ha and 60 Mg/ha. Similar reductions in maple, dogwood, and other tree colonization may occur in forests.
Change in decomposition (rate or equilibrium extent) of native soil organic matter and plant litter	Survey of Microbial processes	±	Effect	A survey of litter decomposition in New Mexico suggests that litter decomposition may increase in the presence of sewage sludge.
Change in balance of processes in nitrogen cycle	Single Chemical Toxicity (Benchmarks)	+		Based on the ORNL benchmark for toxicity of zinc to microbial processes, including nitrification, the addition of sewage sludge may impact the nitrogen cycle.
	Media toxicity tests	±		Based on toxicity tests at higher application rates of sewage sludge, rates of nitrogen fixation, nitrification, and denitrification may be altered.
	Weight of Evidence	+	Effect	Sewage sludge added at 40 Mg/ha is likely to impact the nitrogen cycle in the Douglas-fir forest.

[1] *Risk* indicates an adverse effect; *effect* indicates a change in the community that may or may not be considered adverse.

Based on evidence from media toxicity tests and vegetation surveys, the decomposition of some types of plant litter may accelerate with sludge application. The decomposition rate of native soil organic matter may also be affected, but no test of municipal sewage-sludge-amended media was available.

Based on the ORNL benchmark for the toxicity of zinc to microbial processes, including nitrification, the addition of sewage sludge may impact the nitrogen cycle in the eastern deciduous forest. Toxicity tests at higher application rates than that which was assumed in this assessment have shown that nitrogen fixation, nitrification, and denitrification rates can be altered with sewage sludge amendment. Based on the weight of evidence, sewage sludge added at 40 Mg/ha is likely to impact the nitrogen cycle in the Douglas-fir forest.

4.5.4.4 Southwestern semi-arid rangeland

The weight of evidence for the rangeland is presented in Table 4.66. Based on microbial surveys from Rio Puerco Valley, New Mexico, ammonium oxidizers, total number of bacteria, and population of fungi increase with the addition of sewage sludge. The diversity of fungi are reduced, at least in the short term. Some evidence suggests that the colonization of sagebrush vesicular-arbuscular mycorrhizal fungi may be reduced with application of sewage sludge.

Based on evidence from media toxicity tests and vegetation surveys, the decomposition of some types of plant litter (e.g., blue grama) is likely to accelerate with sludge application. The decomposition rate of native soil organic matter may also be affected, but no test of municipal sewage-sludge-amended media was available.

Based on the ORNL benchmarks for the toxicity of chemicals to microbial processes, metal and organic contaminants in sewage sludge are unlikely to impact the nitrogen cycle in the semi-arid rangeland. Toxicity tests at higher application rates than that which was assumed in this assessment have shown that nitrogen fixation, nitrification, and denitrification rates can be altered with sewage sludge amendment. The impact of sewage sludge added at lower rates is uncertain.

Table 4.66. Weight of evidence for soil microorganisms in the semi-arid rangeland, 1 X 40 Mg/ha application

Endpoint	Evidence	Result	Risk or Effect[1]	Explanation
Change in microbial biomass	Biota Surveys	+	Effect	Total bacteria and total fungi have been observed to increase with sludge application in New Mexico.
Change in microbial diversity	Biota surveys	+	Effect	The diversity of fungi decreased in New Mexico in the short term.
Change in colonization of plants by vesicular-arbuscular mycorrhizal fungi	Biota Surveys	±	Effect	Colonization of sagebrush by VAM fungi is reduced, with the threshold application rate between 30 Mg/ha and 60 Mg/ha. Thus, effects at 40 Mg/ha are uncertain.
Change in decomposition (rate or equilibrium extent) of plant litter	Survey of Microbial processes	+	Effect	A survey of litter decomposition in New Mexico suggests that blue grama litter decomposition increases in the presence of sewage sludge.
Change in balance of processes in nitrogen cycle	Single Chemical Toxicity (Benchmarks)	–		Based on the ORNL benchmarks for toxicity of chemicals to microbial processes, the addition of sewage sludge is unlikely to impact the nitrogen cycle.
	Media toxicity tests	±		Based on toxicity tests at higher application rates of sewage sludge, rates of nitrogen fixation, nitrification, and denitrification may be altered.
	Biota surveys	+		Biomass of ammonium oxidizers have been observed to increase with sludge application in New Mexico
	Weight of Evidence	±		The impact of sewage sludge added at 40 Mg/ha to the nitrogen cycle in the semi-arid rangeland is uncertain.

[1] *Risk* indicates an adverse effect; *effect* indicates a change in the community that may or may not be considered adverse.

4.5.5 Uncertainties

- **Background concentrations.** Background concentrations of chemicals in soil were often responsible for a substantial fraction of the estimated toxicity in sludge-treated ecosystems. Background levels varied significantly among the ecosystems in this study. The spatial variability of background levels within an ecosystem and that within sludge-treated ecosystems in the United States have not been incorporated in this assessment. The likelihood of mobilization of background levels of chemicals by the application of sewage sludge is unknown.

- **Incorporation of sewage sludge.** The rate of incorporation of chemicals in sewage sludge into soils is unknown, as is the conservatism of the assumption that the sewage sludge is mixed in the surface 15 cm of soil.

- **Bioavailability of Contaminants.** The potential difference in bioavailability between chemicals in sludge-treated soil and chemicals in untreated soil has been discussed previously. It is also notable that bioavailability may vary from sludge to sludge and from soil to soil. Toxicity tests rarely came from the particular assessment ecosystems for which they were used.

- **Decomposition of sludge and bioavailability of chemicals.** The bioavailability of metals and other chemicals applied to soils in sewage sludge may be slowly altered with the decomposition of organic matter in sludge.

- **Extrapolation of ambient media toxicity test data.** It has been assumed that media toxicity tests and biota surveys at application rates of 40 Mg/ha are comparable to the assumptions in this assessment. To determine if this is true, the concentrations of active chemicals (particular nutrients or contaminants) in sludge would have to be measured, as well as the ultimate concentrations in soil.

- **Single contaminant tests vs exposure to multiple contaminants in the field.** While organisms in the four ecosystems are exposed to multiple contaminants concurrently, published toxicological values from which ORNL benchmarks were derived only consider effects experienced from exposures to single contaminants. Because some contaminants to which microorganisms are exposed can interact antagonistically, single contaminant studies may overestimate their toxic potential. Similarly, for those contaminants that interact additively or synergistically, single contaminant studies may underestimate their toxic potential.

- **Inorganic forms or species present in the environment.** Toxicity of metal species varies dramatically depending upon the valence state or form (organic or inorganic) of the metal. Relative toxicity to microbial processes are generally unknown. The NSSS does not report species or form of contaminants in municipal sewage sludges.

5. CONCLUSIONS

As stated in the problem formulation chapter, observed effects of sewage sludge on biota or ecological processes may result from nutrients or non-nutrient chemicals in sewage sludge or a combination of the two. Predictions of effects based on the toxicity of single metals or nitrogen alone may not be accurate when effects of all of the components of sewage sludge act together. Predictions of impacts to several endpoints in this assessment rely on information about either nutrients or contaminants but not both. For example, net primary productivity in the three forests was estimated with information on nitrogen and carbon alone. Impacts on wildlife were predicted based on metal or organic chemical data, though it is acknowledged that wildlife would be impacted by changes in forage quality, changes in litter habitat, and changes in populations of food items. The weights of evidence for plant and microbial endpoints are attempts to integrate the impacts from nutrients and contaminants.

It must be emphasized that the assessment and conclusions focus on a narrowly-defined scenario: 95% UCL on the mean chemical concentrations from the NSSS, 40 Mg/ha application rate, sewage sludge mixed in the surface 15 cm of soil. An exception is the nitrogen modeling of production, which examined multiple scenarios, including multiple applications. The assessment findings are only valid in this context.

5.1 Plant Productivity and Leaching of Nitrate

A significant gain in forest productivity is usually associated with application of sewage sludge. The increase is attributed to the addition of nitrogen, which is widely limiting in many forest soils. All three forests have shown increased tree growth both in actual field applications of biosolids and in modeling scenarios, though the eastern deciduous forest has responded to a lesser degree than the Douglas-fir or loblolly pine plantations. Annual applications above about 400 kg N/ha generally do not show further benefits in productivity and can lead to detrimental effects due to nitrate contamination of groundwater. As ecosystem capacity for nitrogen utilization is exceeded, the mineralization of organic N to nitrate is not matched by nitrate uptake and immobilization within the ecosystem. All three forests have shown increased nitrate leaching with applications of biosolids that exceed ecosystem retention capacity.

Nitrate levels in soils of biosolids-treated land tend to reach soil solution concentrations of untreated land within three years after treatment. Biosolids applications at levels that do not exceed ecosystem capacity for N retention, repeated at 3 to 4 year intervals, should provide increased ecosystem productivity without significant nitrate contamination of groundwater.

Increased forest productivity with biosolids application is generally accompanied by higher soil organic matter content. This effect leads to improved soil fertility and increased turnover and availability of nutrients needed for plant growth.

In the deciduous forest simulations, differences in species sensitivity to N stress and shading lead to shifts in biomass proportions within the community. Greater overstory dominance may suppress, but not eliminate, the growth of understory species, according to simulation results.

In the New Mexico rangeland, it has been observed that increased productivity associated with sludge amendment may be followed by increased drought stress in these plants. The importance and extent of this phenomenon are unknown.

The impacts of biosolids on natural ecosystems may involve risks that need not be taken if the supply of biosolids is less than the demand that could be developed by commercial forest plantations. For example, if there are only enough sources of biosolids to meet 10% of the N needed to increase productivity of forest plantations, preference should be given to biosolids application on commercial forests rather than in unmanaged or natural ecosystems.

Based on empirical evidence and the depth to groundwater, nitrate leaching from sewage sludge-amended soils in rangeland is not expected to impact groundwater. Based on nitrogen additions, increased productivity is likely in the rangeland with minimal amendments.

5.2 Other Impacts of Sewage Sludge on Plant Communities

Sewage sludge may present a hazard to plant populations in the four ecosystems, based on zinc toxicity. The finding of potential risk is based on estimated concentrations of zinc in plant tissues and soil in the years following sludge application; long-term measurements are not available to confirm or refute this conclusion. The issue of short-term toxic effects due to salinity is raised in the rangeland ecosystem, and it is unknown whether this is an equivalent potential problem in the forests.

Understory composition and biomass are likely to change in forests, but whether the change constitutes a risk is dependent on the management goals for the forest or plantation. Similarly, changes in plant cover, biomass, species diversity, forage quality, and perhaps noxious species are likely to change in ways that may benefit range management. These benefits may be weighed against potential risks from metal toxicity and salinity.

5.3 Impacts of Sewage Sludge on Wildlife

The conclusions of the weight of evidence are that individual fox, shrews, American robins, and western meadow larks are not at risk from a one-time application of 40 Mg/ha of sewage sludge (containing contaminant concentrations reported in the NSSS) in any ecosystem, and individual deer are not at risk in the Douglas-fir forest, Loblolly pine plantation, or semiarid rangeland ecosystems. Limited bioaccumulation data suggests that white-tailed deer in the eastern deciduous forest may be at risk from copper and zinc. Because populations consist of individuals and individual-level risks are unlikely, population-level risks to all endpoints considered are also unlikely.

In addition to potentially toxic contaminants, sewage sludge contains plant nutrients which have been shown to increase plant growth and alter plant community composition and structure. These effects of application of sewage sludge on plants can also have indirect effects on the wildlife communities present at sludge treated sites. Due to the minimal extent of contaminant risks estimated or observed at sludge-treated sites and because wildlife responses to plant and habitat-mediated effects of sludge application have been observed, it is likely that the plant and habitat-mediated effects of sludge application will predominate.

5.4 Impacts of Sewage Sludge on Soil Invertebrates

Soil invertebrates in the Douglas-fir forest, eastern deciduous forest, and loblolly pine plantations are unlikely to be adversely affected by metal or organic contaminants in sewage sludge-treated soil. Effects of nutrients in sludge on the balance of populations (e.g., earthworms, mites) in the soil invertebrate community are possible and are observed at higher application rates than that assumed here. Impacts of sewage sludge on the dominant invertebrates in the semi-arid rangeland are unknown.

5.5 Impacts of Sewage Sludge on Soil Microbial Processes

The total biomass of microorganisms is likely to increase with sludge application. Litter decomposition in all ecosystems is likely to accelerate with sludge application; this may constitute an adverse effect, depending on the management goals of the ecosystem. The effect may be reduced through the use of de-watered sludge, spread in large clumps. The balance between nitrification and denitrification and nitrogen fixation by cyanobacteria and rhizobia has been showed to be altered and reduced, respectively, at application rates higher than that assumed in this study. Nitrification may be reduced by the zinc component of sewage sludge. Changes in the balance of processes in the nitrogen cycle are likely in the Douglas-fir forest and in the eastern deciduous forest and are more uncertain in the loblolly pine plantation and in the rangeland.

5.6 Regulatory Levels of Contaminants

Existing cumulative loading limits for individual elements in sewage sludge were not screened in this study. There is substantial uncertainty associated with estimates of the quantity of elements that remain in surface soils after a number of years (or for different periods of time in the case of multiple applications), though mass balances are beginning to be performed at some locations (e.g., McBride et al. 1997). The bioavailability of elements that were applied in sewage sludge to soils decades ago is not easily estimated. An ecological risk assessment of cumulative loading limits for the application of municipal sewage sludge in forests and rangeland would not be very definitive at this time.

A risk assessor could attempt to estimate protective cumulative loading limits based on multiple lines of evidence (single chemical toxicity, ambient media toxicity, and field surveys), but such estimates would also not be definitive. These lines of evidence come from different ecosystems, soils, sludges, application rates, and organisms, and any estimate of protective loading limits would not be very precise. It is recommended that additional research be performed to support such a risk assessment (Sect. 6).

6. FUTURE RESEARCH

Further research is needed to reduce the uncertainties associated with this ecological risk assessment as well as to fill critical data gaps. The following is an unprioritized list of research needs.

- Long term monitoring of sewage sludge-treated sites is needed to determine changes in bioavailability of metals and other added chemicals with the decomposition of organic matter added in sewage sludge. Mass balances should be performed.

- Extraction methods should be developed that represent the bioavailability of particular contaminants to key organisms, such as plants, earthworms, and microorganisms.

- The relative bioavailability of metals in sludge-treated soils versus untreated soils to invertebrates and microorganisms should be investigated.

- Research is needed to reduce the uncertainty in uptake models. Including soil characteristics as a variable in regressions would probably improve the predictability of the models.

- Soil-to-plant uptake factors were rarely derived from studies from the specific ecosystems of concern. If the Douglas-fir forest, eastern deciduous forest, loblolly pine plantations, and semi-arid rangeland continue to represent ecosystems where sewage sludge is likely to be applied, critical plants in the ecosystems should be identified, and soil and plant tissue concentrations should be measured using those species.

- Research is needed on the uptake and toxicity of chemicals in sewage sludge by soil invertebrates in the semiarid rangeland.

- Given the high uptake of sludge-borne metals to arthropods, particularly those feeding on plants, and the observed toxicity to aphids, spiders, and cabbage loopers at high sludge application, toxicity thresholds of sewage sludge application to arthropods should be investigated. A highly exposed group of herbivores would be those that feed on plant roots.

- The role of sewage sludge in altering insect abundance, which could alter herbivory of trees or food availability to wildlife has not been studied. The fertilization of black locust trees has previously been shown to alter insect herbivory (Hargrove et al. 1984).

- Little is known about the toxicity of molybdenum to plants, with or without sludge amendments.

- It is likely that litter decomposition accelerates in the presence of sewage sludge. The extent of this effect, the impacts of the reduced litter habitat on wildlife and litter fauna, and the impacts of different management strategies should be investigated in long-term experiments. For example, it is likely that de-watered sewage sludge would lead to isolated rather than uniform regions of rapid litter decomposition.

- Long-term and short-term studies are needed to investigate the impacts of sewage sludge on microbial processes at common application rates, given the reduction in nitrogen fixation above 100 Mg/ha.

- Impacts of sewage sludge on mycorrhizal fungi, both vesicular-arbuscular and ectomycorrhizal fungi need to be studied in each of the key ecosystems where they are important. Moreover, if their populations are reduced with sludge amendment, the survival and growth of plants that ordinarily require them, particularly in drought conditions, should be assessed.

- The level of background metals in soils often contributed a large fraction of the estimated concentrations in soils, following the application of sewage sludge. The role of this portion of the total metal concentration in toxicity and uptake by biota should be investigated.

- Salinity measurements following sewage sludge addition are rarely taken, except in the semi-arid rangeland. Monitoring of salinity in soils and research on the acute toxicity of salinity in combination with the nutrients, metals, and organic matrix of sewage sludge are needed.

- Research is needed to predict effects of directly impacted populations (e.g., increases in earthworms, increases in microbial biomass, decreases in insectivorous birds) with other populations and processes in the ecosystems.

- Research is needed to compare the bioavailability of metals applied to soils in multiple versus single applications of sewage sludge. No evidence has been found to support different uptake or toxicity of metals from a single application of 40 Mg/ha versus ten applications of 4 Mg/ha.

- Surveys of invertebrates in ecosystems where de-watered sewage sludge has been applied are lacking.

- The LINKAGES model would be useful for predicting changes in the Douglas-fir forest community. Research is needed to refine the parameterization of the model for the dominant and secondary tree species in this ecosystem.

- A major need in the modeling of production and forest community changes is revision of the decomposition and litter algorithms to account for effects of biosolids. It has been shown that forest litter decomposes more readily in the presence of biosolids. Some coding and algorithm development would be needed to account for biosolids effects on litter and decomposition process.

- Research is needed to improve the sewage sludge/humic matter decomposition subroutine of the LINKAGES program. Biosolids should be treated as a separate compartment from the humic matter and litter, with a different decomposition rate.

- Research is needed to ascertain the value that people place on particular ecosystem conditions. For example, the long-term changes predicted in the forest composition of Hubbard Brook from sludge application may or may not constitute a risk, depending on the value placed on the possible species mixes.

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FIGURES

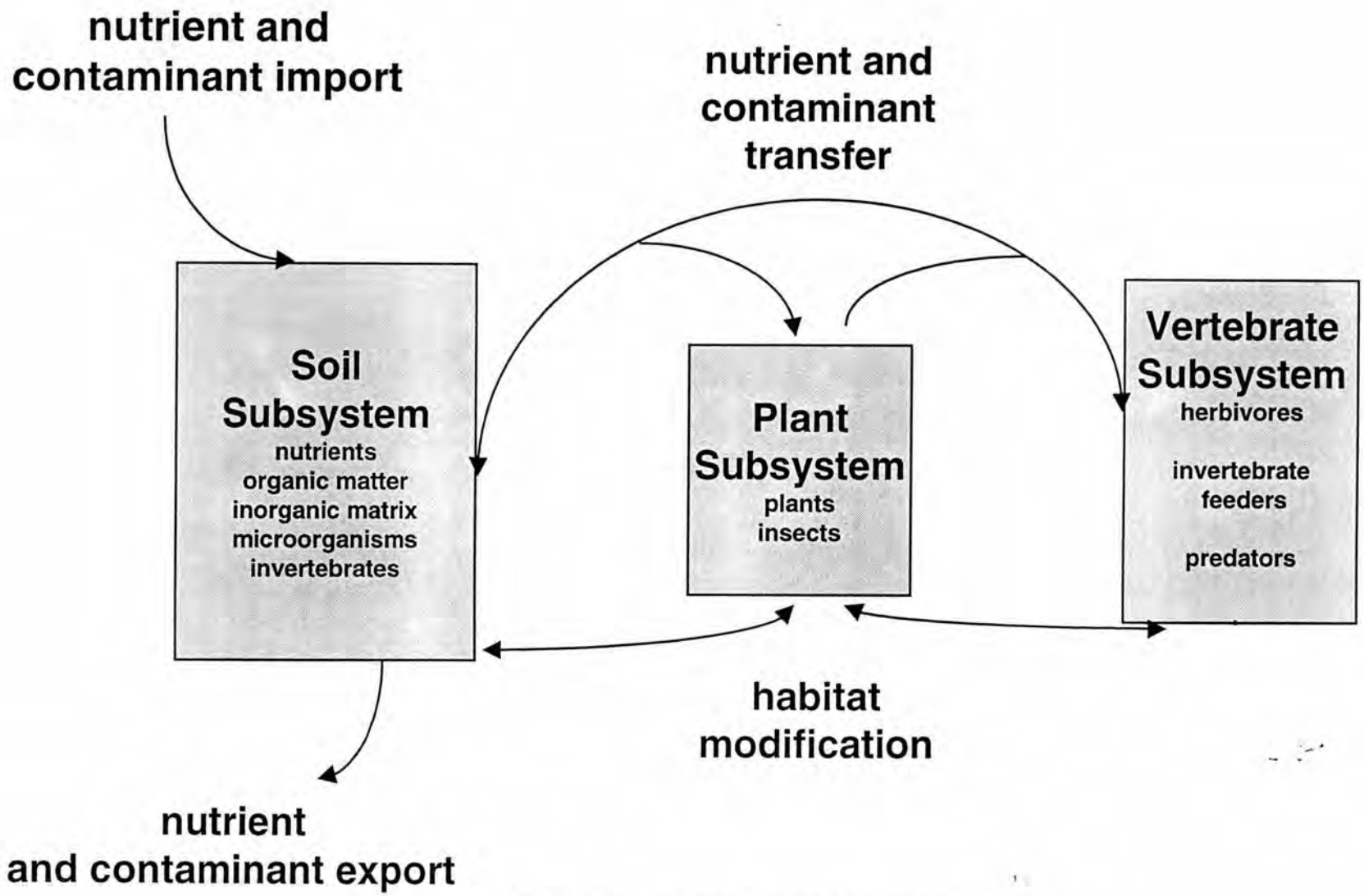

Fig. 1.1. Effects of biosolids on terrestrial ecosystems

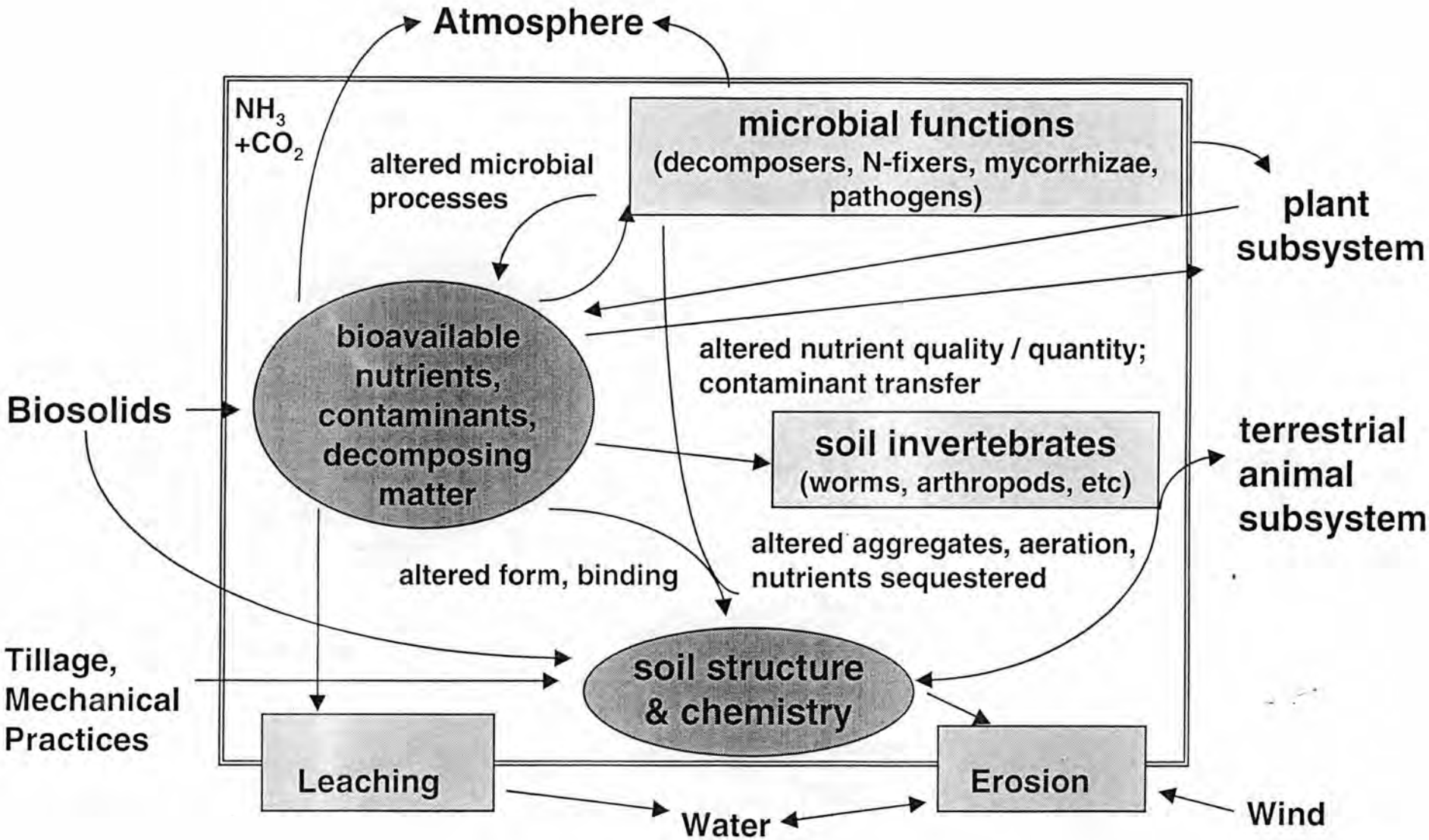

Fig. 1.2. Effects of biosolids on the soil subsystem

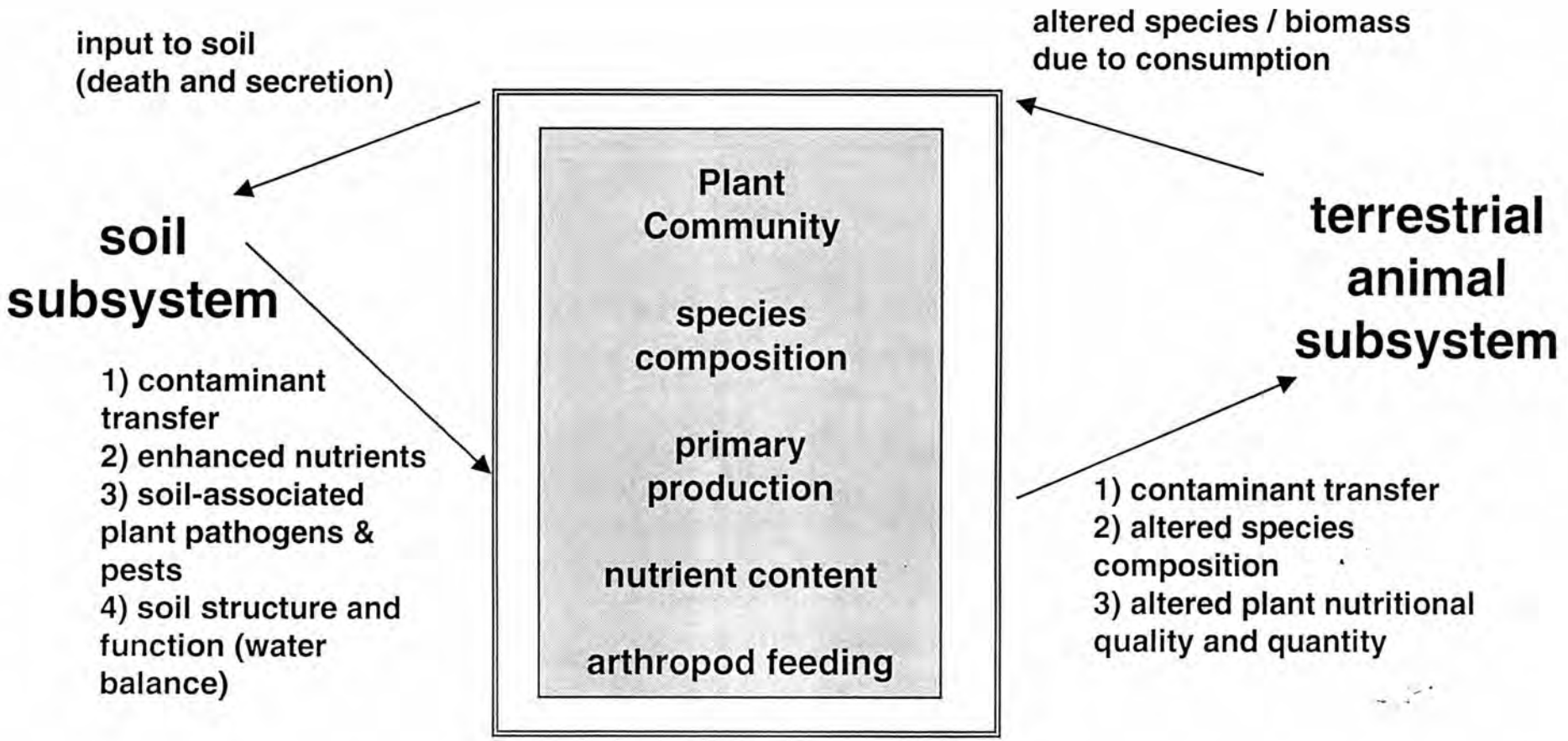

Fig. 1.3. Effects of biosolids on the plant subsystem

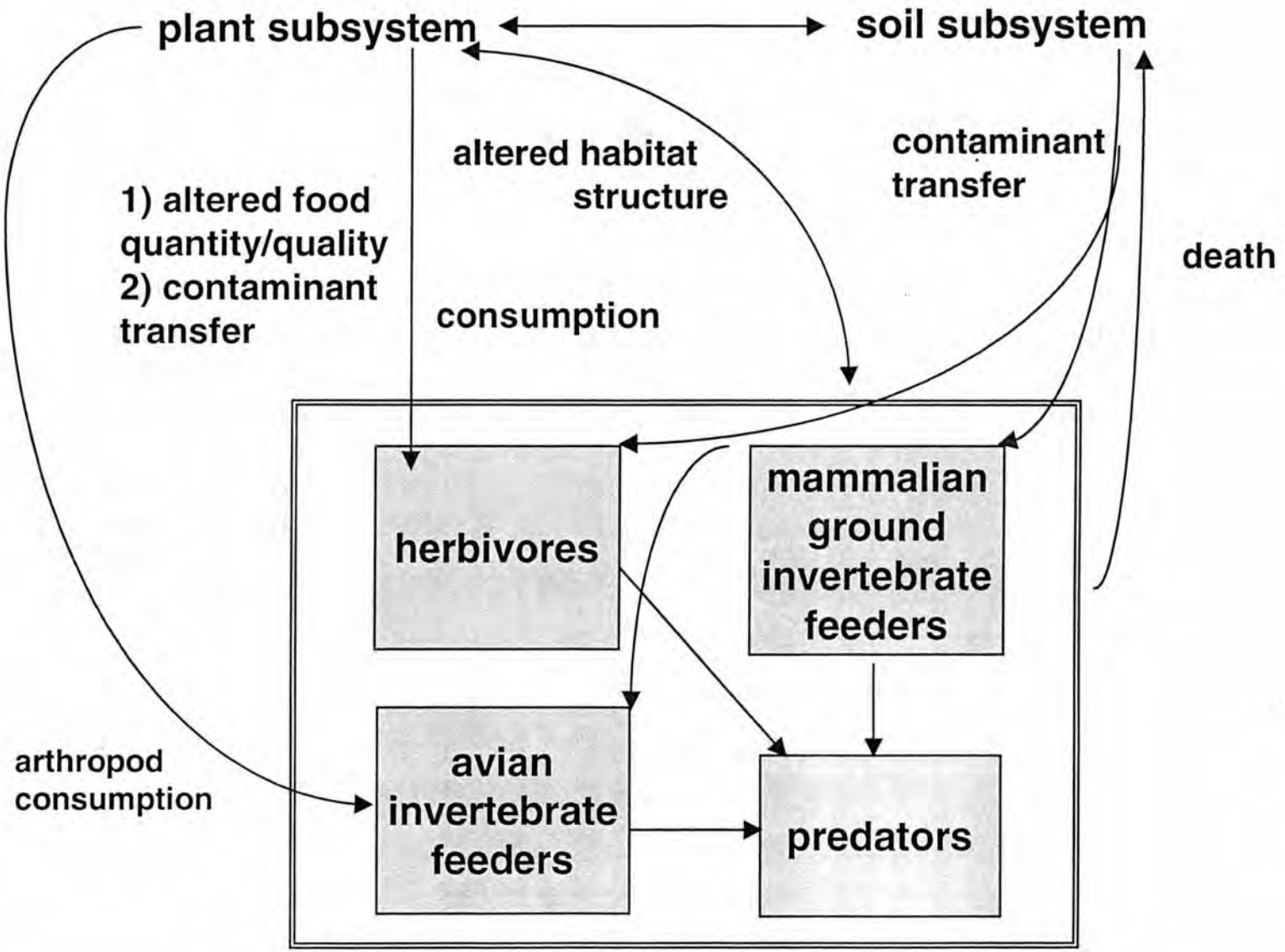

Fig. 1.4. Effects of biosolids on the terrestrial animal subsystem

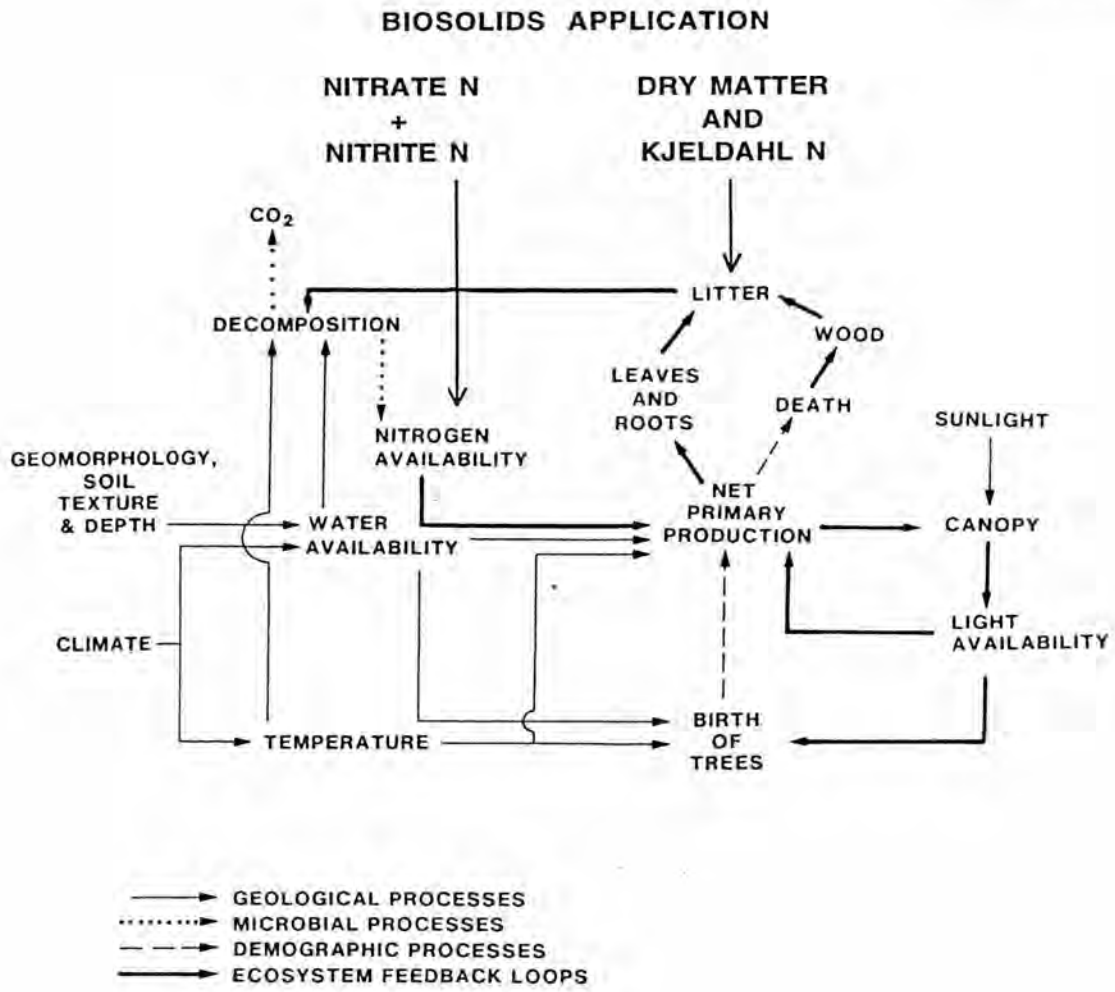

Fig. 2.1. Hypothesized feedbacks and constraints in forest ecosystems (the LINKAGES model).

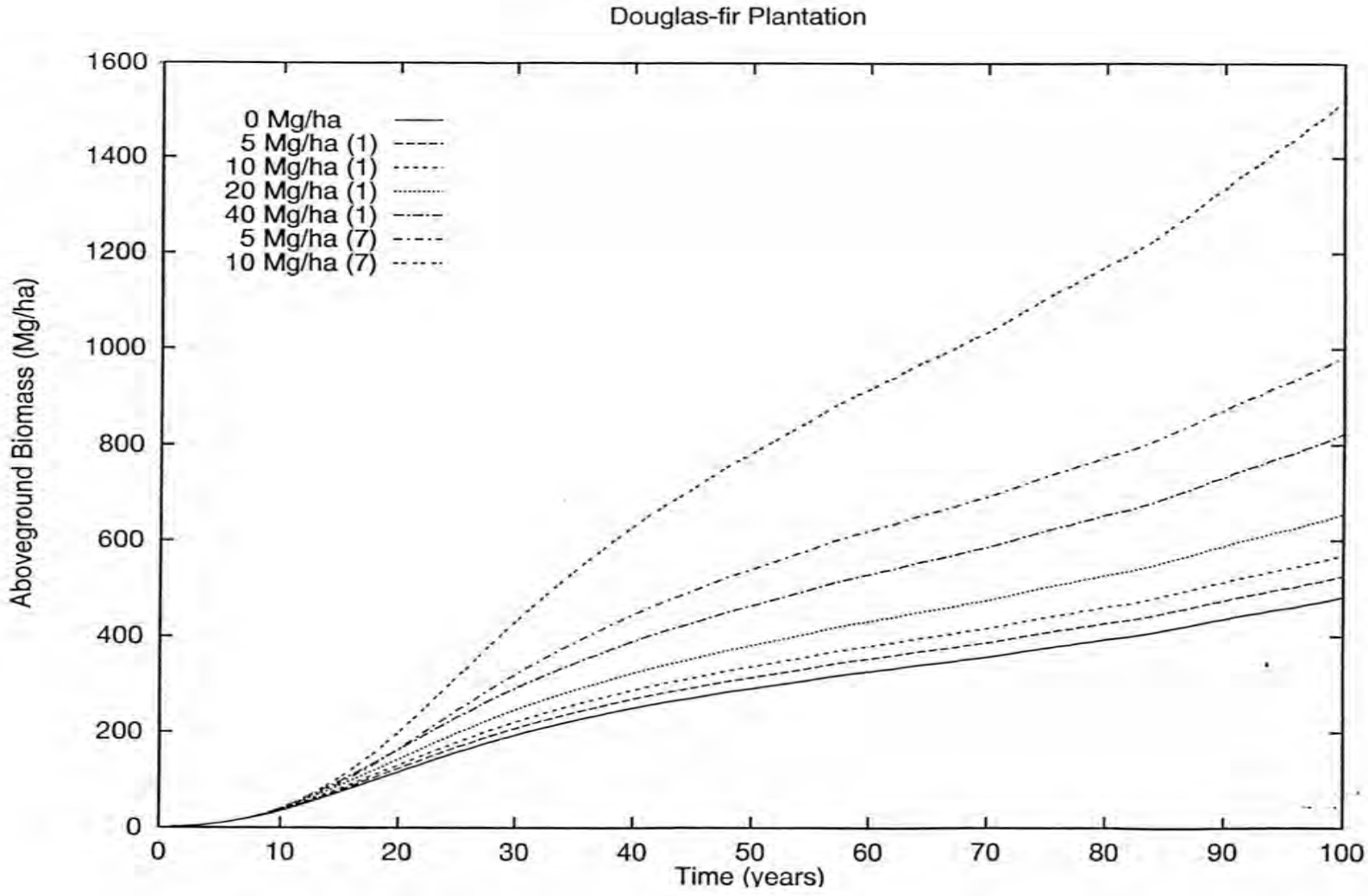

Fig. 2.2. Simulation of 100 years of aboveground biomass of Douglas-fir under six sludge application scenarios and no sludge treatment.

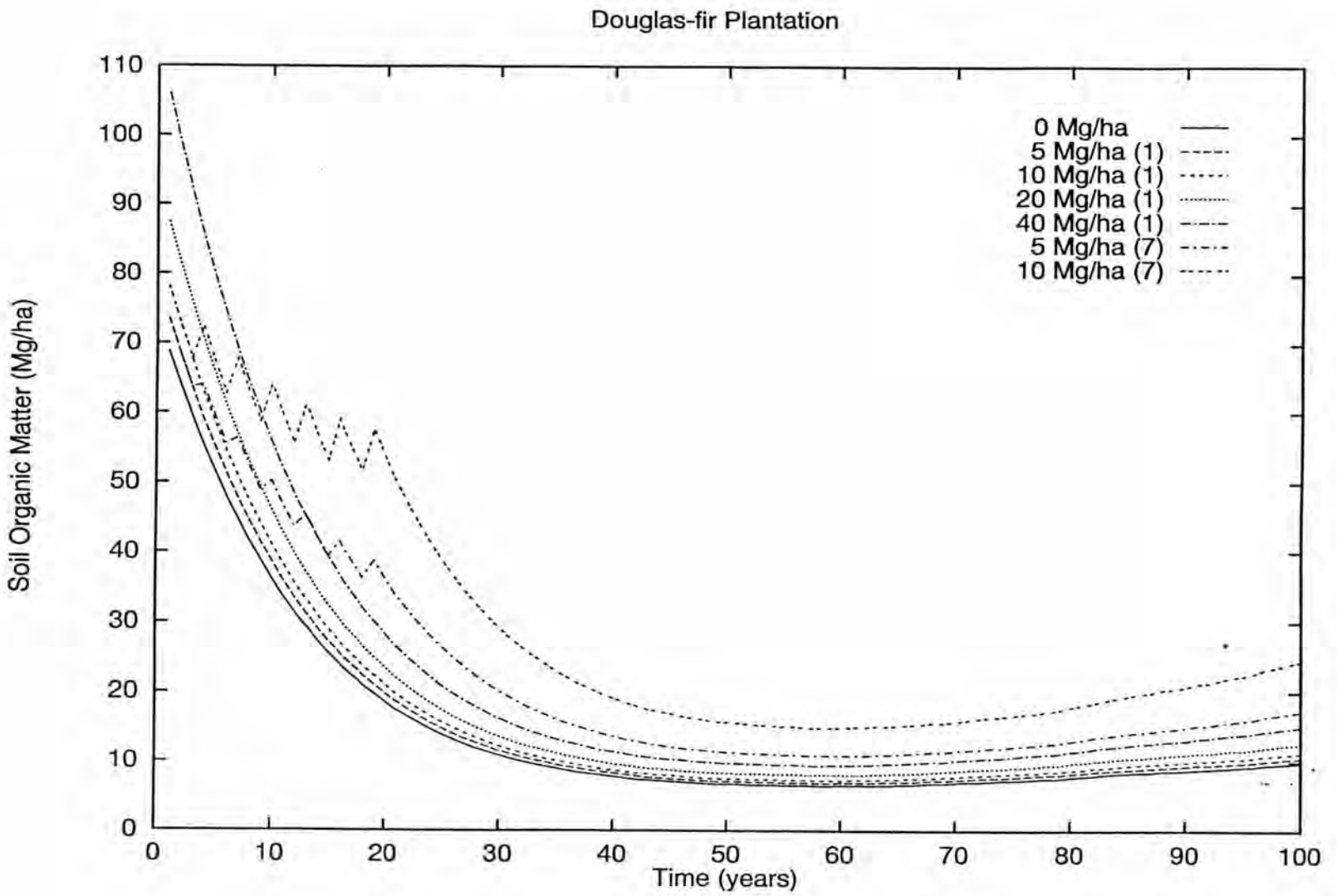

Fig. 2.3. Simulation of 100 years of soil organic matter in a Douglas-fir plantation under six sludge application scenarios and no sludge treatment.

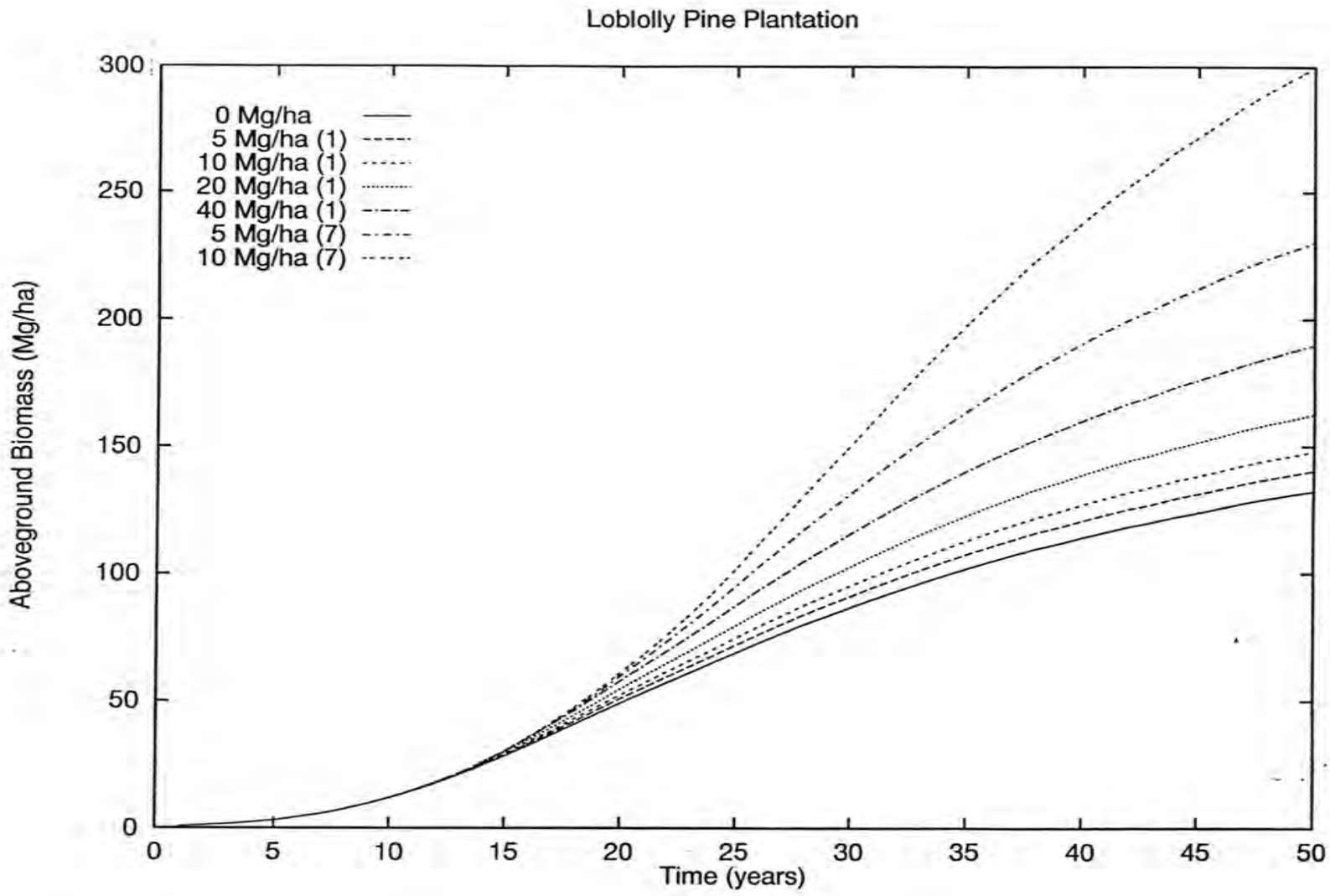

Fig. 2.4. Simulation of 100 years of aboveground biomass of loblolly pine under six sludge application scenarios and no sludge treatment.

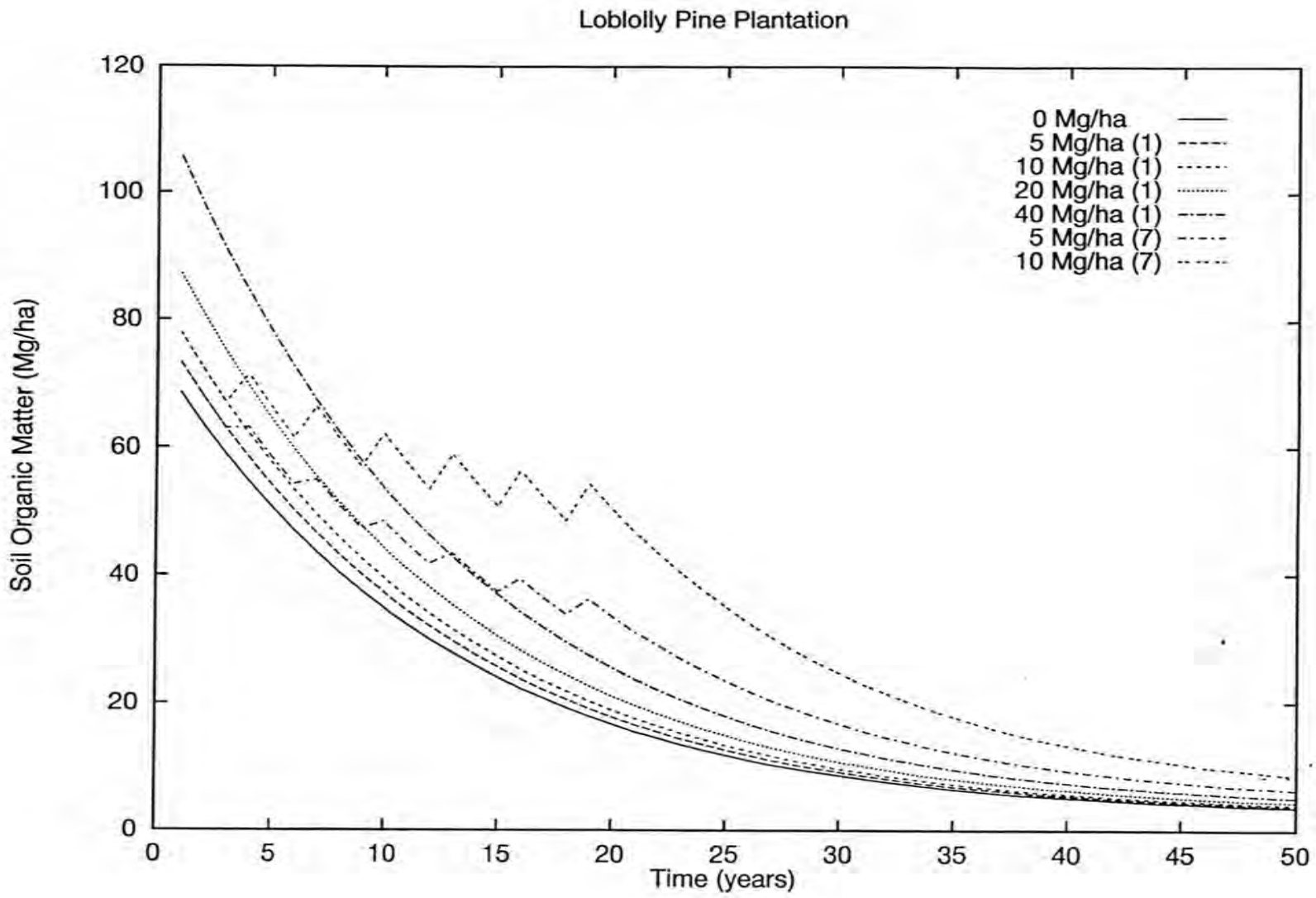

Fig. 2.5. Simulation of 100 years of soil organic matter in a loblolly pine plantation under six sludge application scenarios and no sludge treatment.

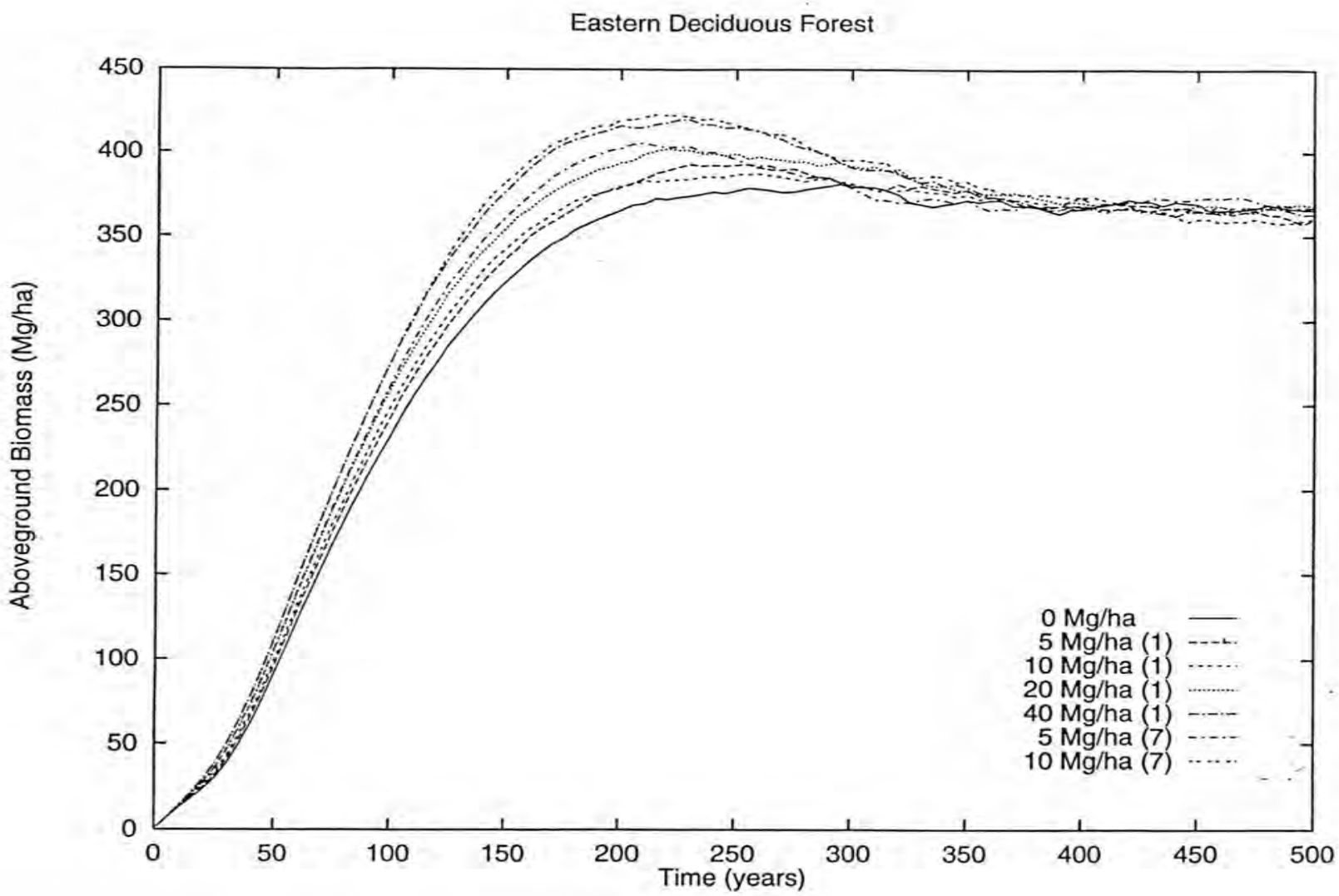

Fig. 2.6. Simulation of 100 years of aboveground biomass of trees in an eastern deciduous forest under six sludge application scenarios and no sludge treatment.

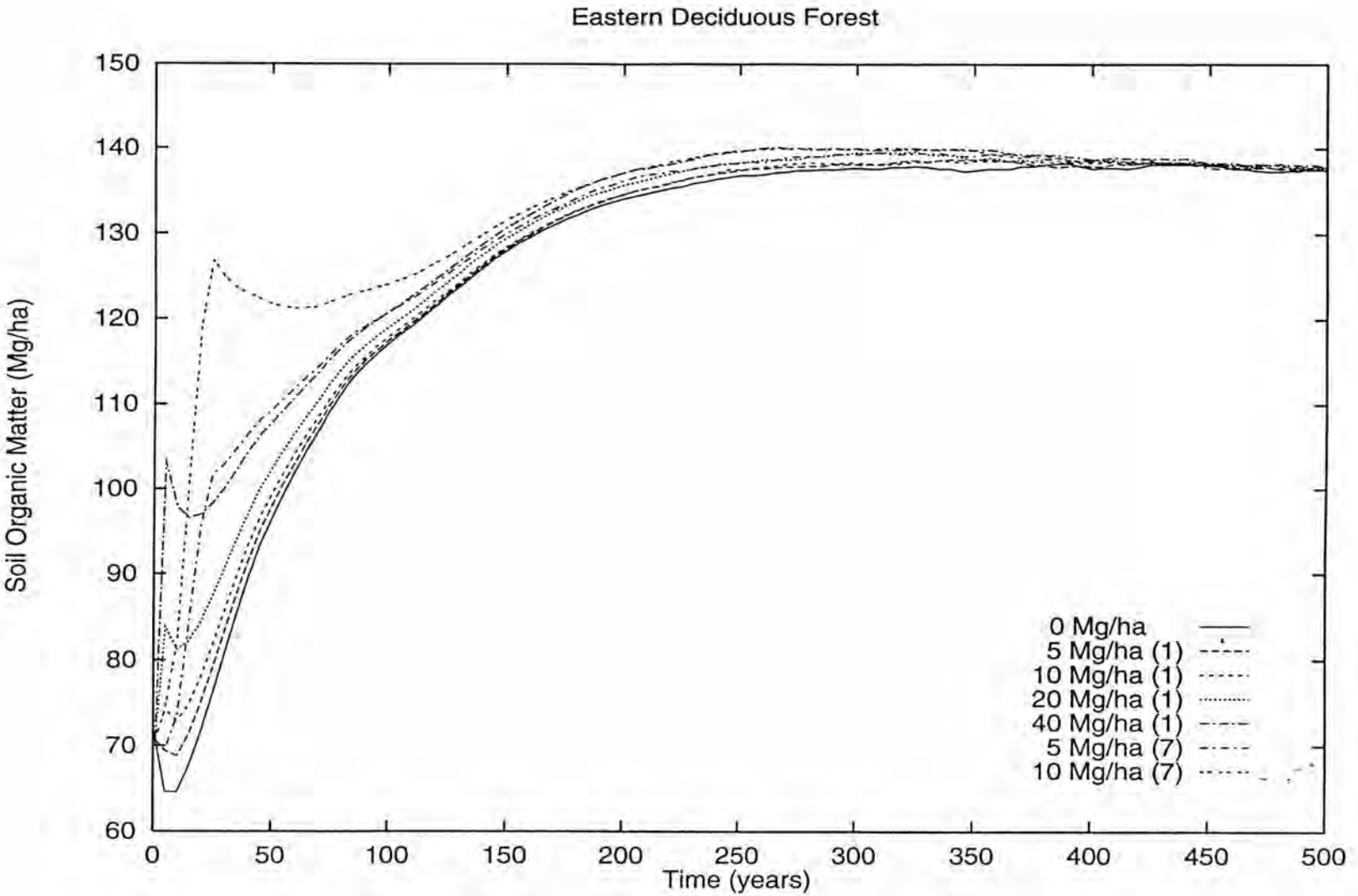

Fig. 2.7. Simulation of 100 years of soil organic matter in an eastern deciduous forest under six sludge application scenarios and no sludge treatment.

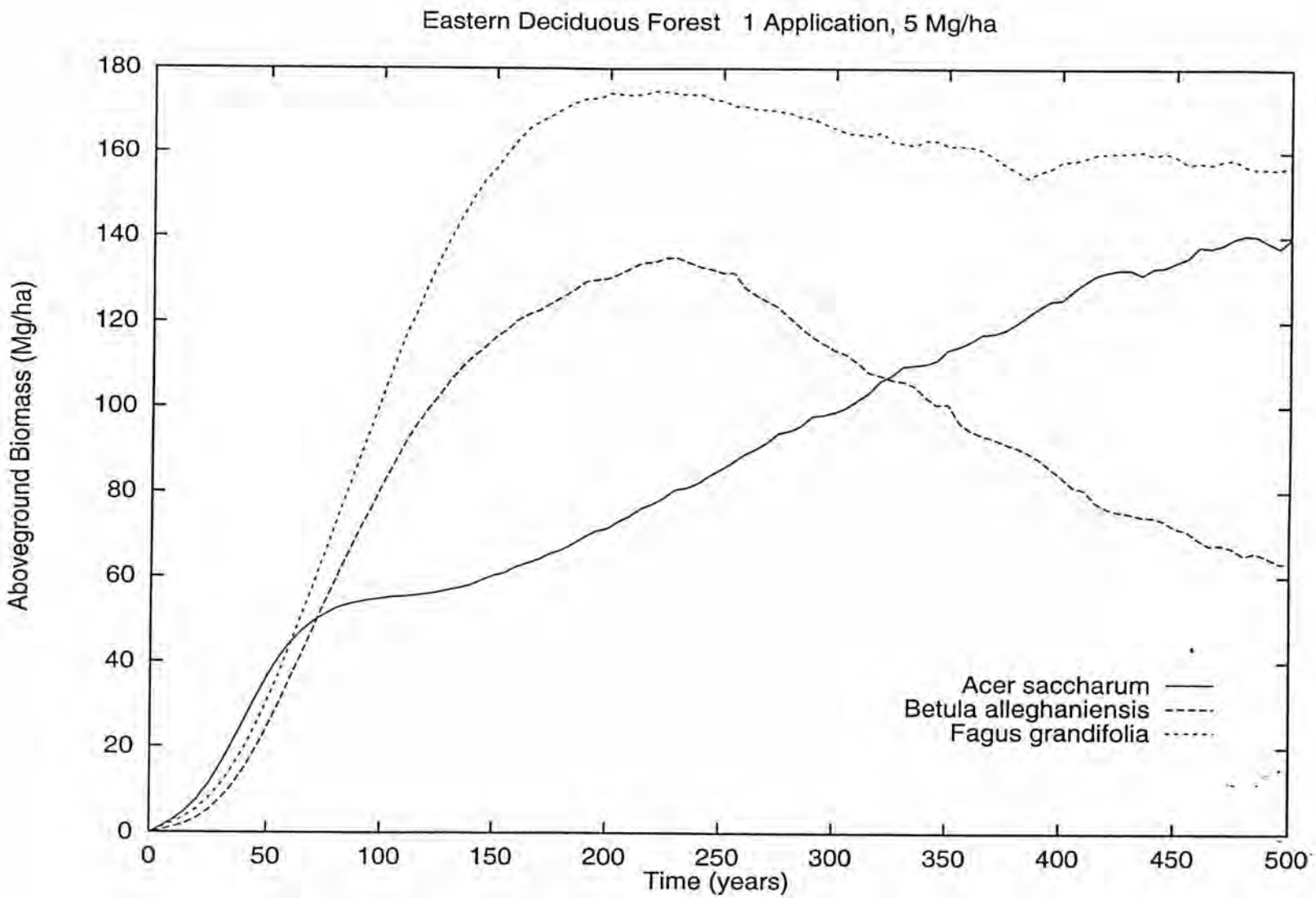

Fig. 2.8a. 50-year simulation of aboveground biomass of three dominant eastern deciduous forest tree species at Hubbard Brook, NH, if sludge is applied once at 5 Mg/ha.

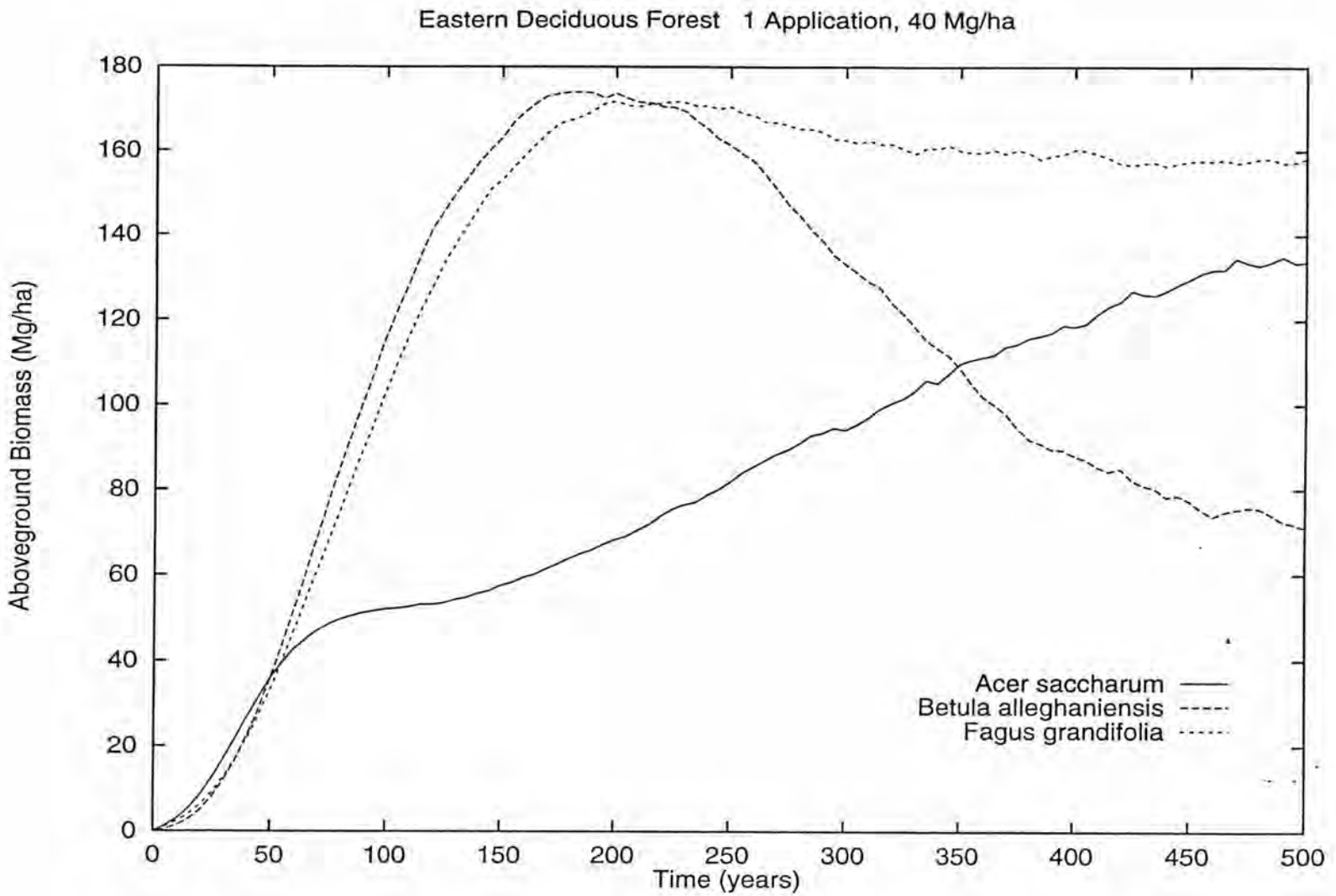

Fig. 2.8b. 50-year simulation of aboveground biomass of three dominant eastern deciduous forest tree species at Hubbard Brook, NH, if sludge is applied once at 40 Mg/ha.

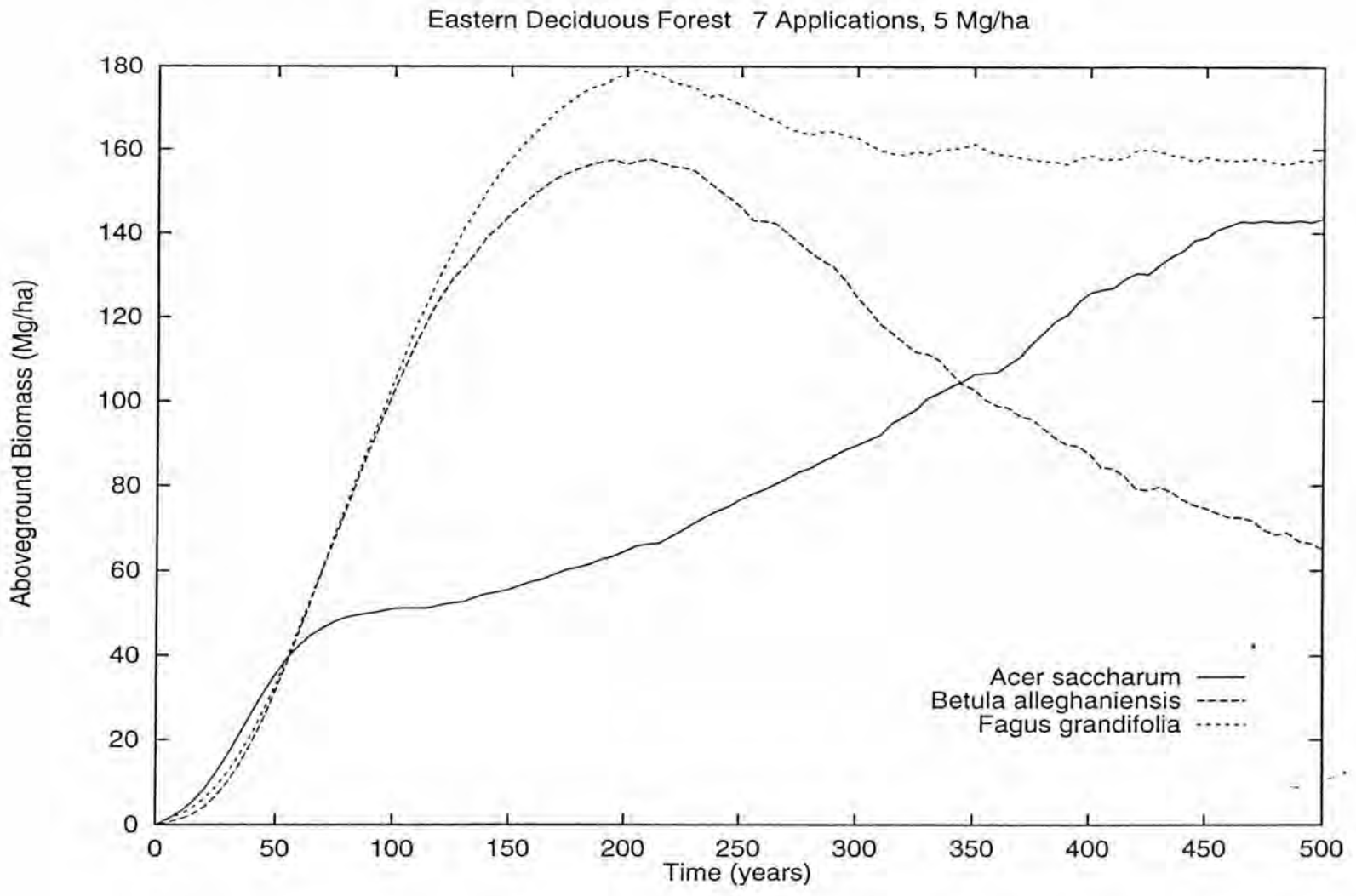

Fig. 2.8c. 50-year simulation of aboveground biomass of three dominant eastern deciduous forest tree species at Hubbard Brook, NH, if sludge is applied seven times (every three years) at 5 Mg/ha.

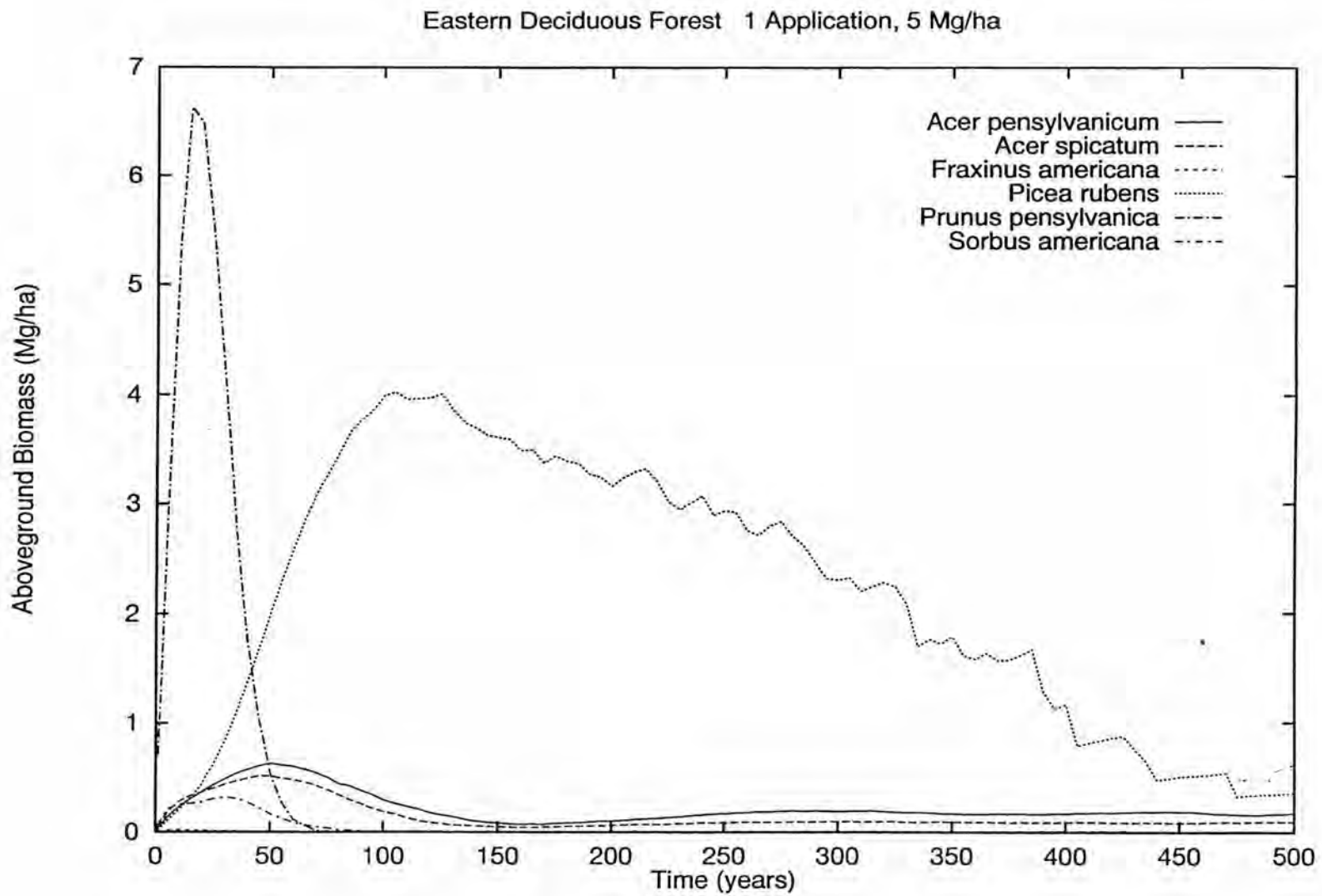

Fig. 2.9a. 500-year simulation of aboveground biomass of six eastern deciduous forest tree species at Hubbard Brook, NH, if sludge is applied once at 5 Mg/ha.

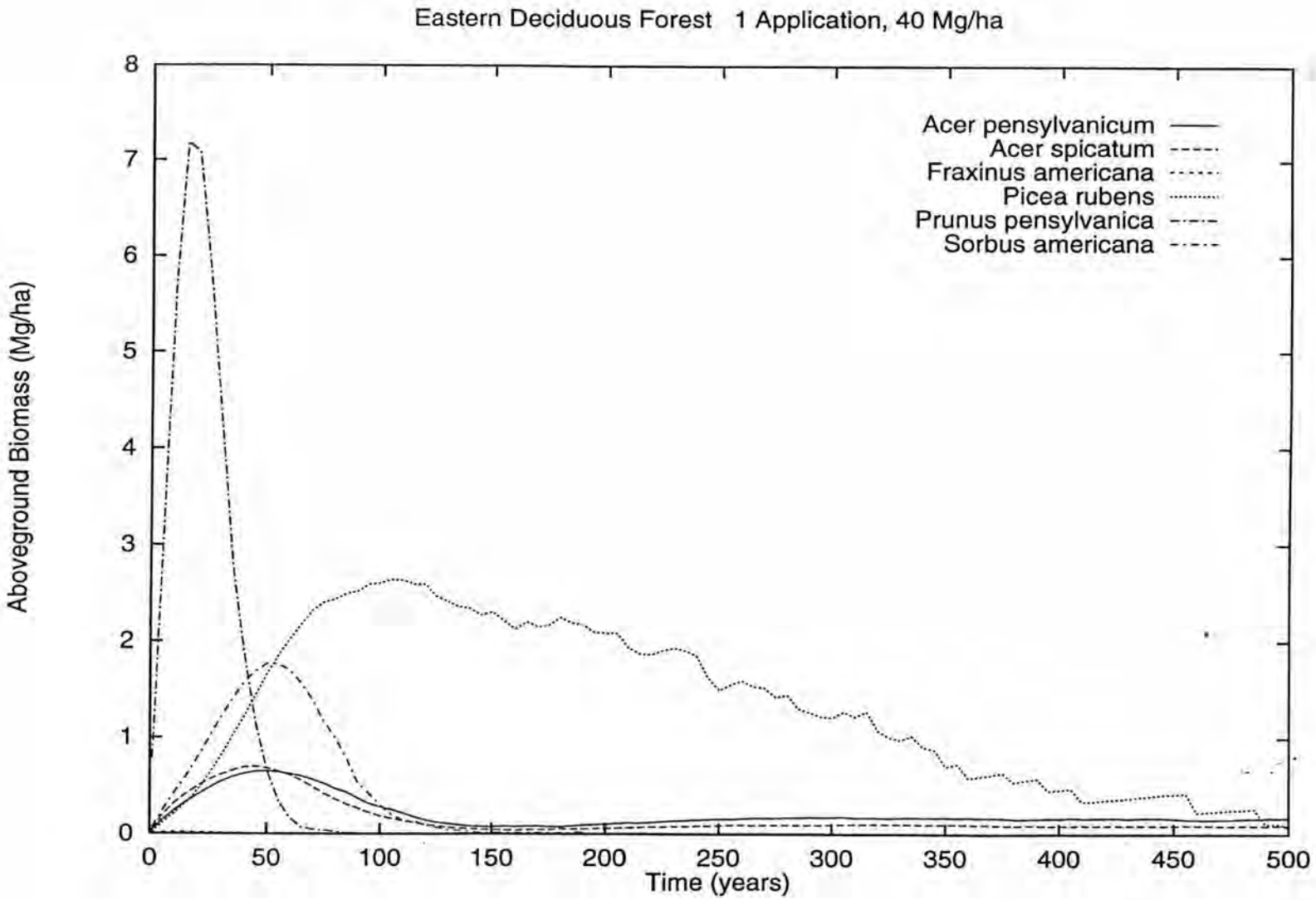

Fig. 2.9b. 500-year simulation of aboveground biomass of six eastern deciduous forest tree species at Hubbard Brook, NH, if sludge is applied once at 40 Mg/ha.

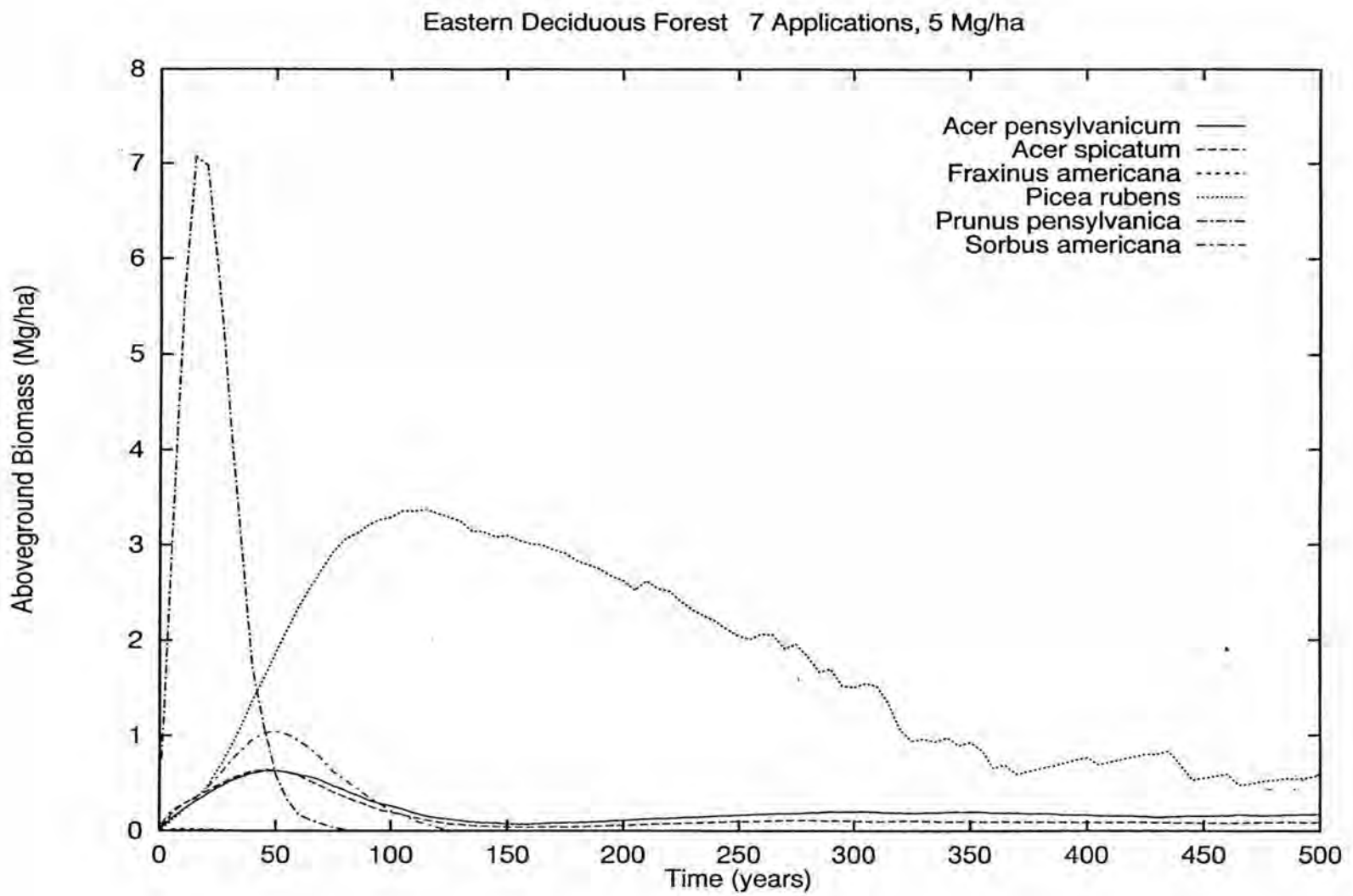

Fig. 2.9c. 500-year simulation of aboveground biomass of six eastern deciduous forest tree species at Hubbard Brook, NH, if sludge is applied seven times (every three years) at 5 Mg/ha.

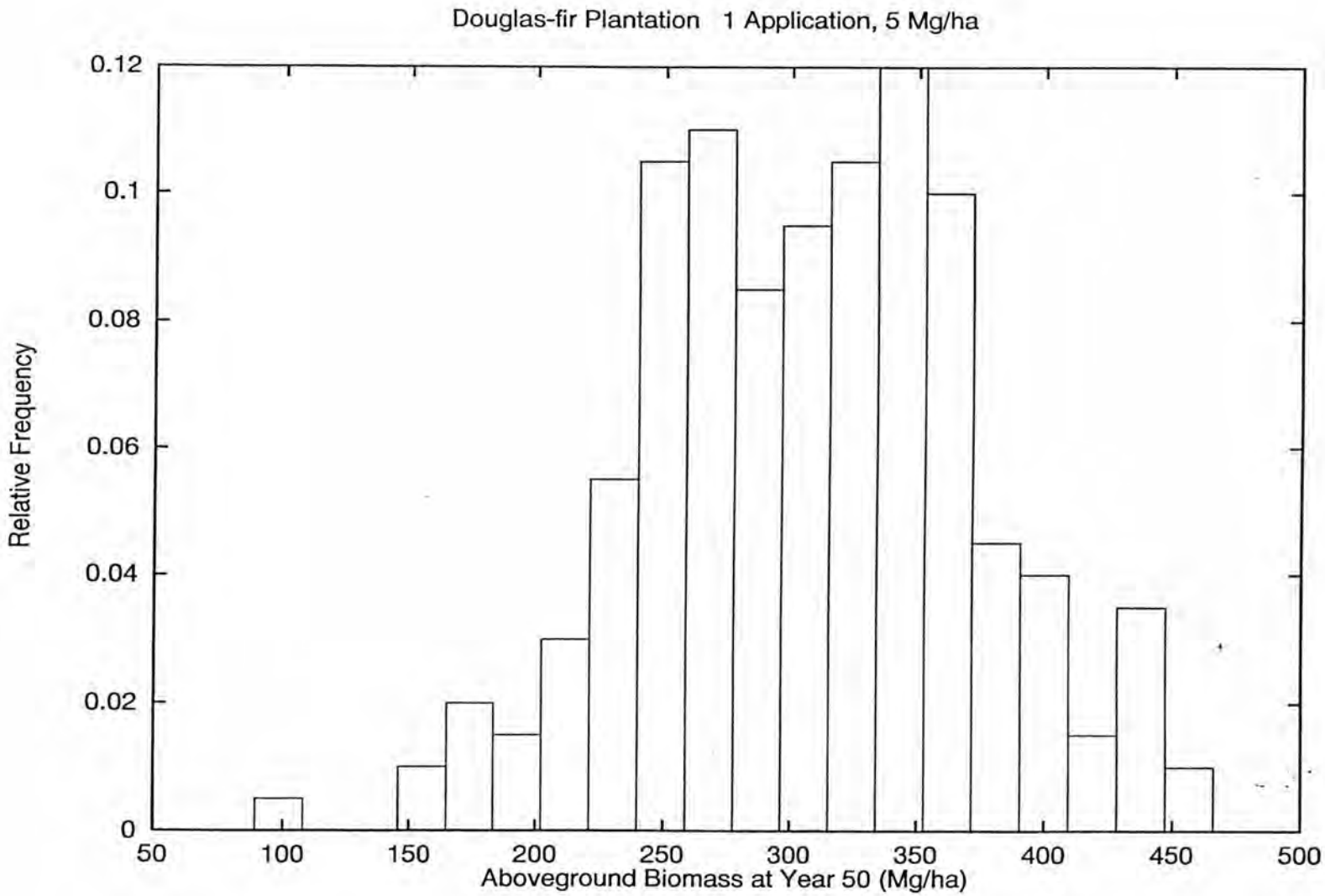

Fig. 2.10a. Frequency distribution of aboveground biomass of Douglas-fir at year 50 following a single application of sludge at 5 Mg/ha.

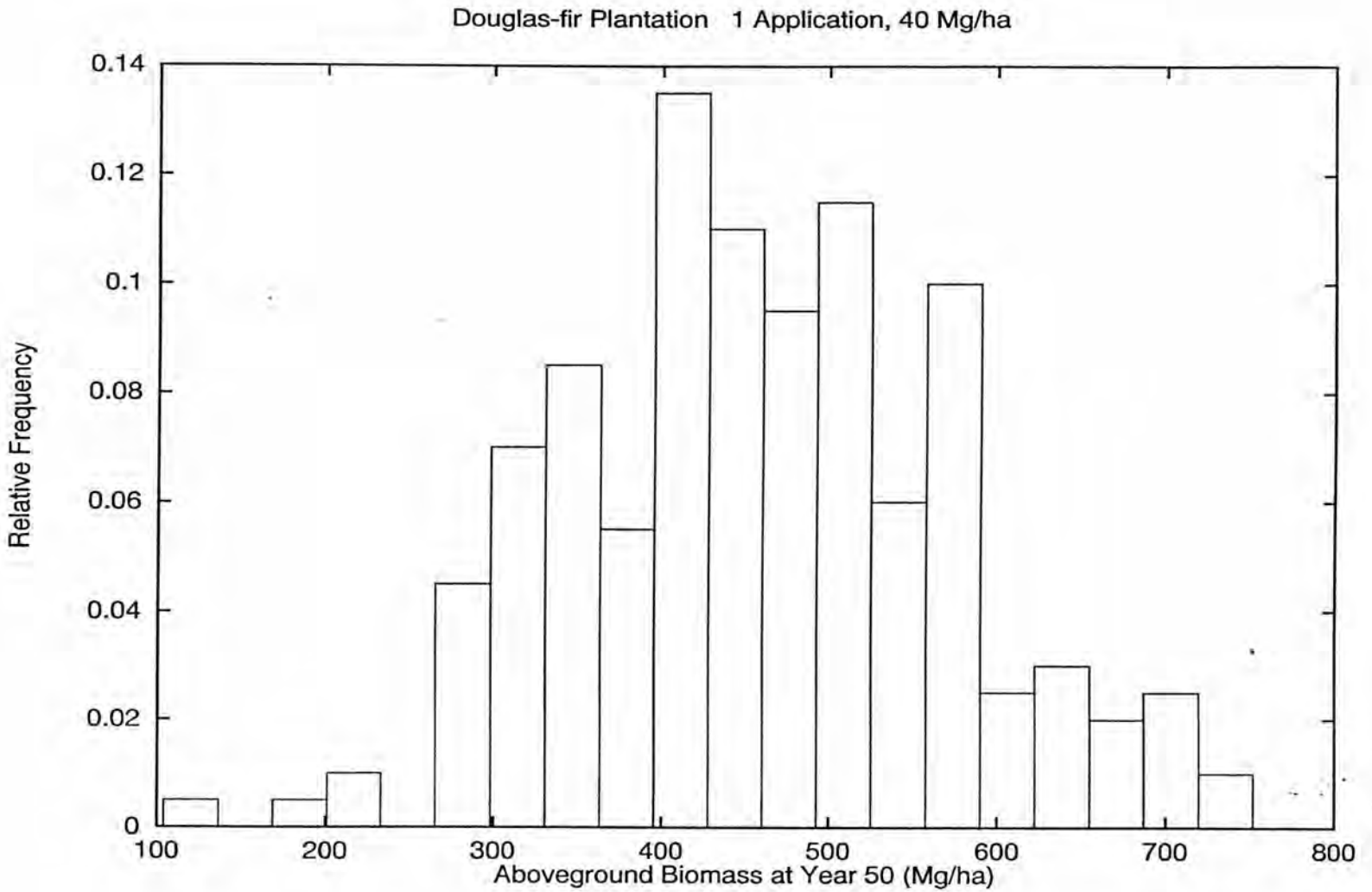

Fig. 2.10b. Frequency distribution of aboveground biomass of Douglas-fir at year 50 following a single application of sludge at 40 Mg/ha.

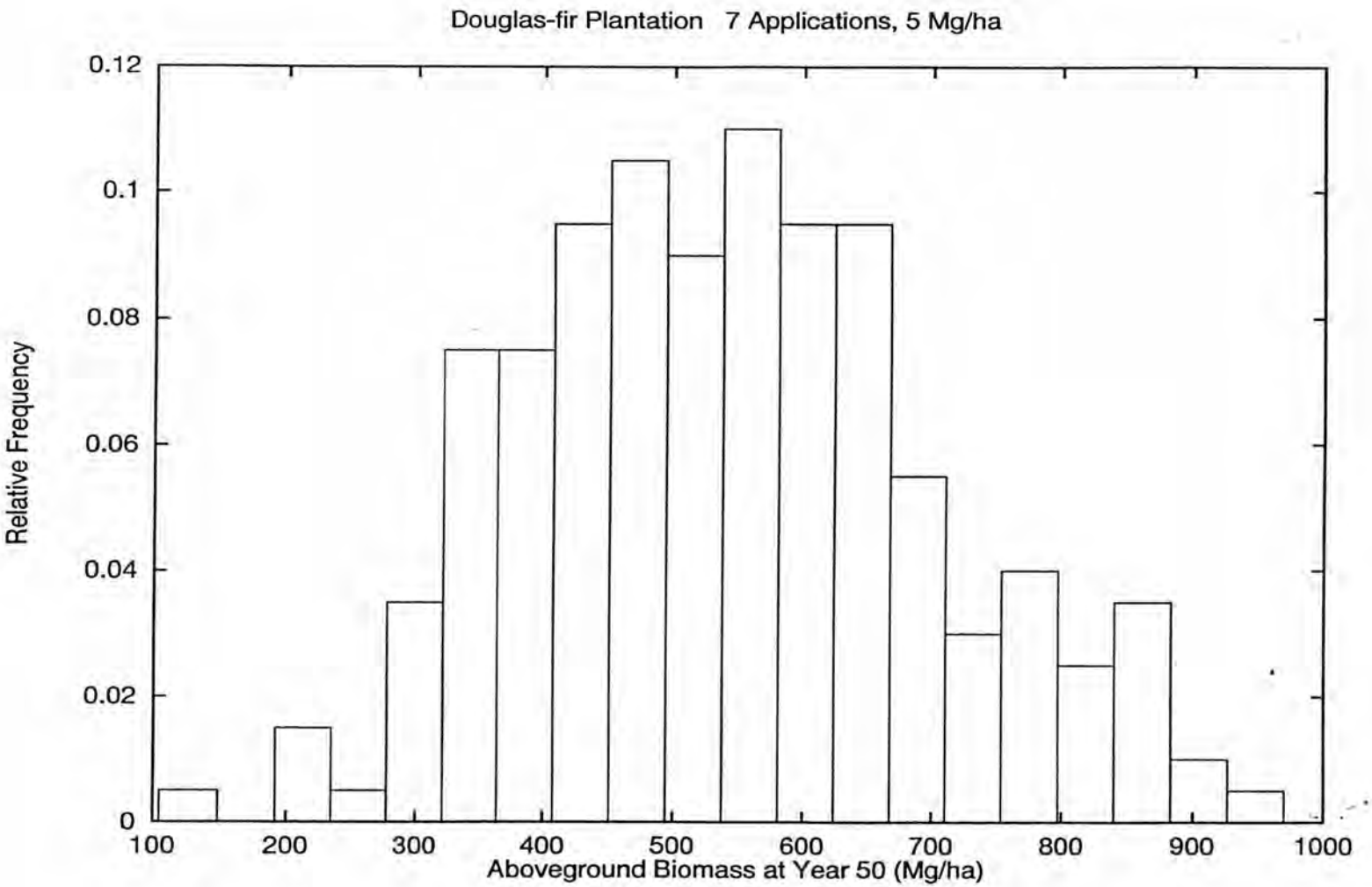

Fig. 2.10c. Frequency distribution of aboveground biomass of Douglas-fir at year 50 following seven applications of sludge at 5 Mg/ha, every three years.

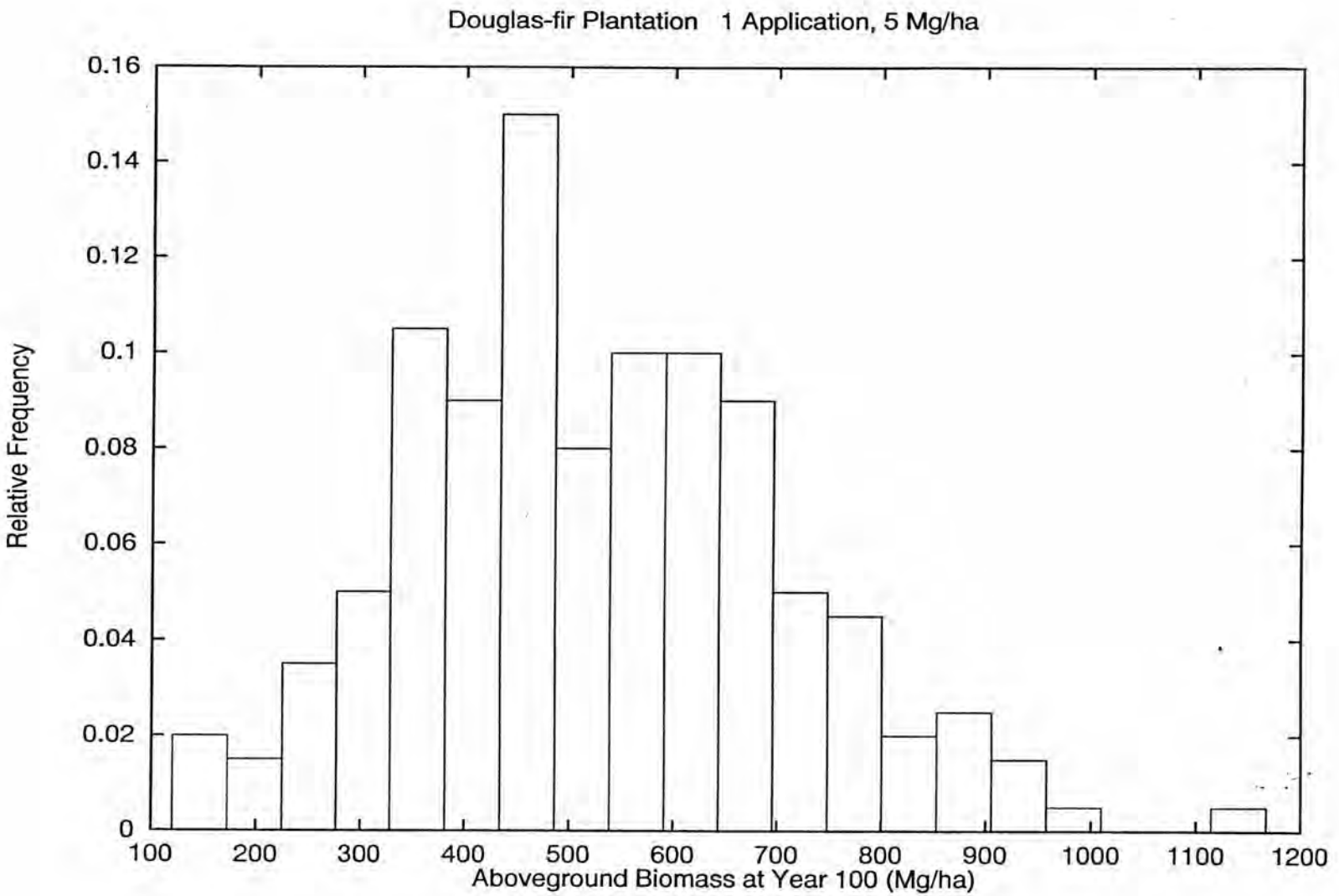

Fig. 2.11a. Frequency distribution of aboveground biomass of Douglas-fir at year 100 following a single application of sludge at 5 Mg/ha.

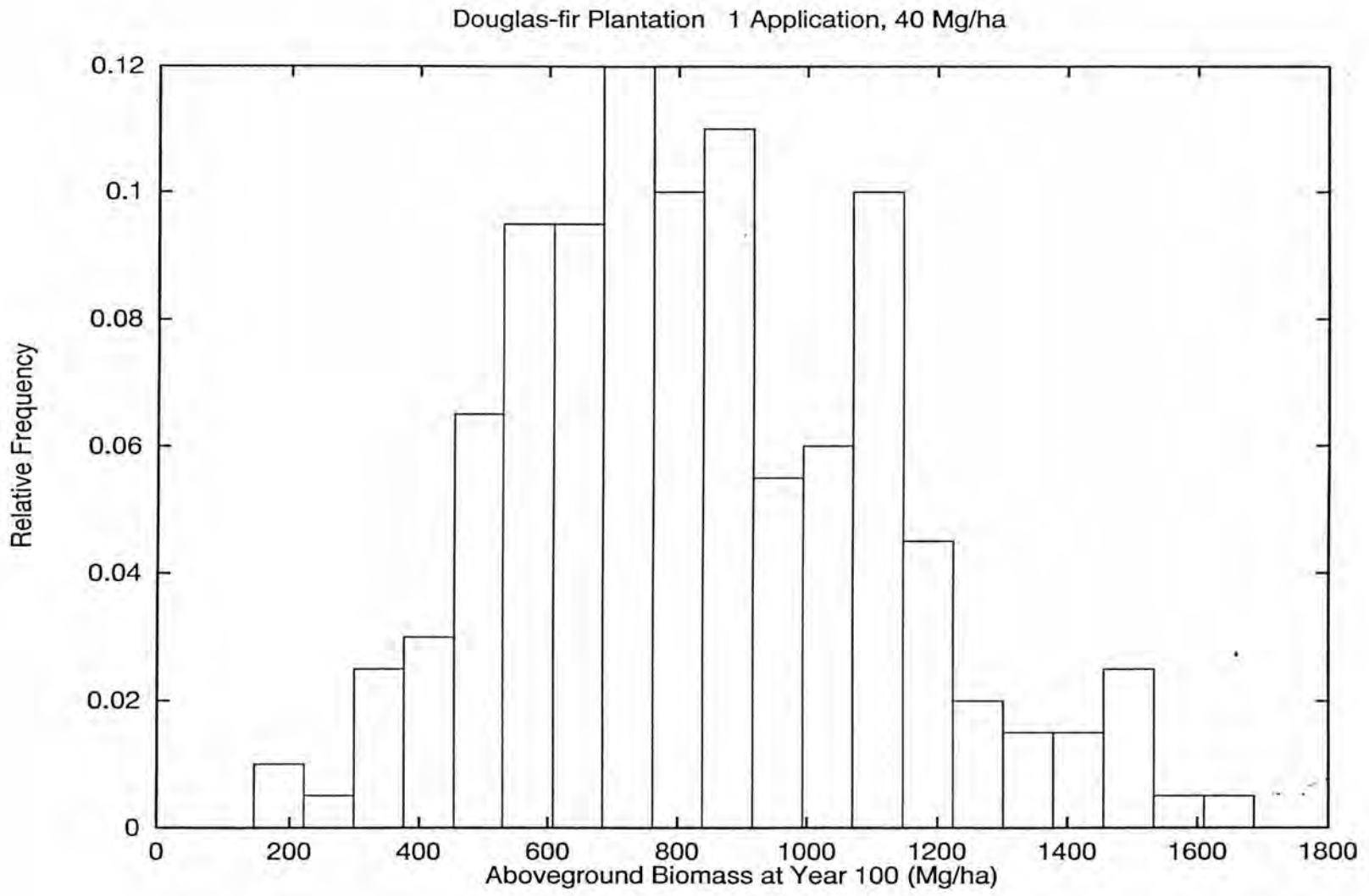

Fig. 2.11b. Frequency distribution of aboveground biomass of Douglas-fir at year 100 following a single application of sludge at 40 Mg/ha.

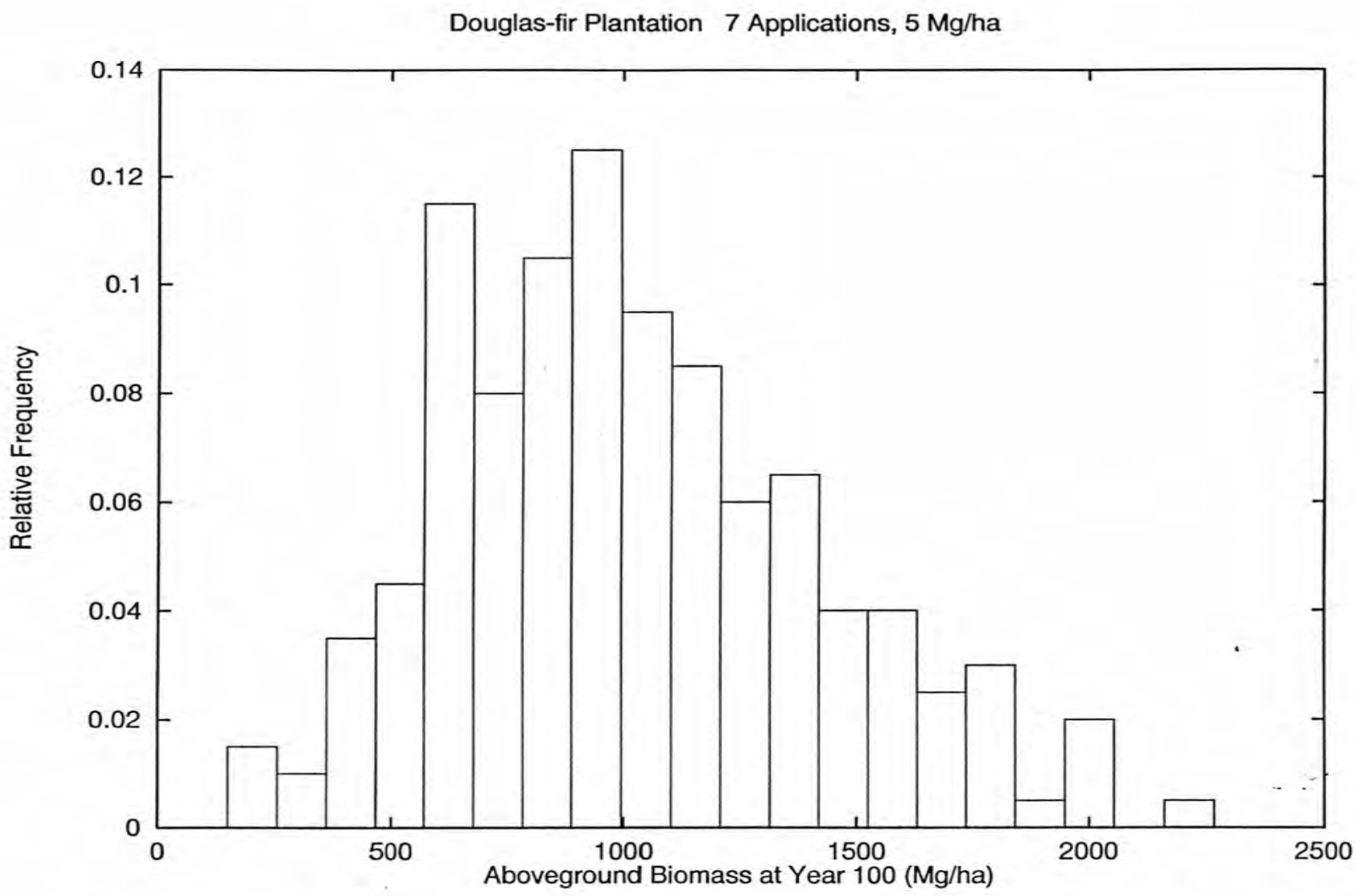

Fig. 2.11c. Frequency distribution of aboveground biomass of Douglas-fir at year 100 following seven applications of sludge at 5 Mg/ha, every three years.

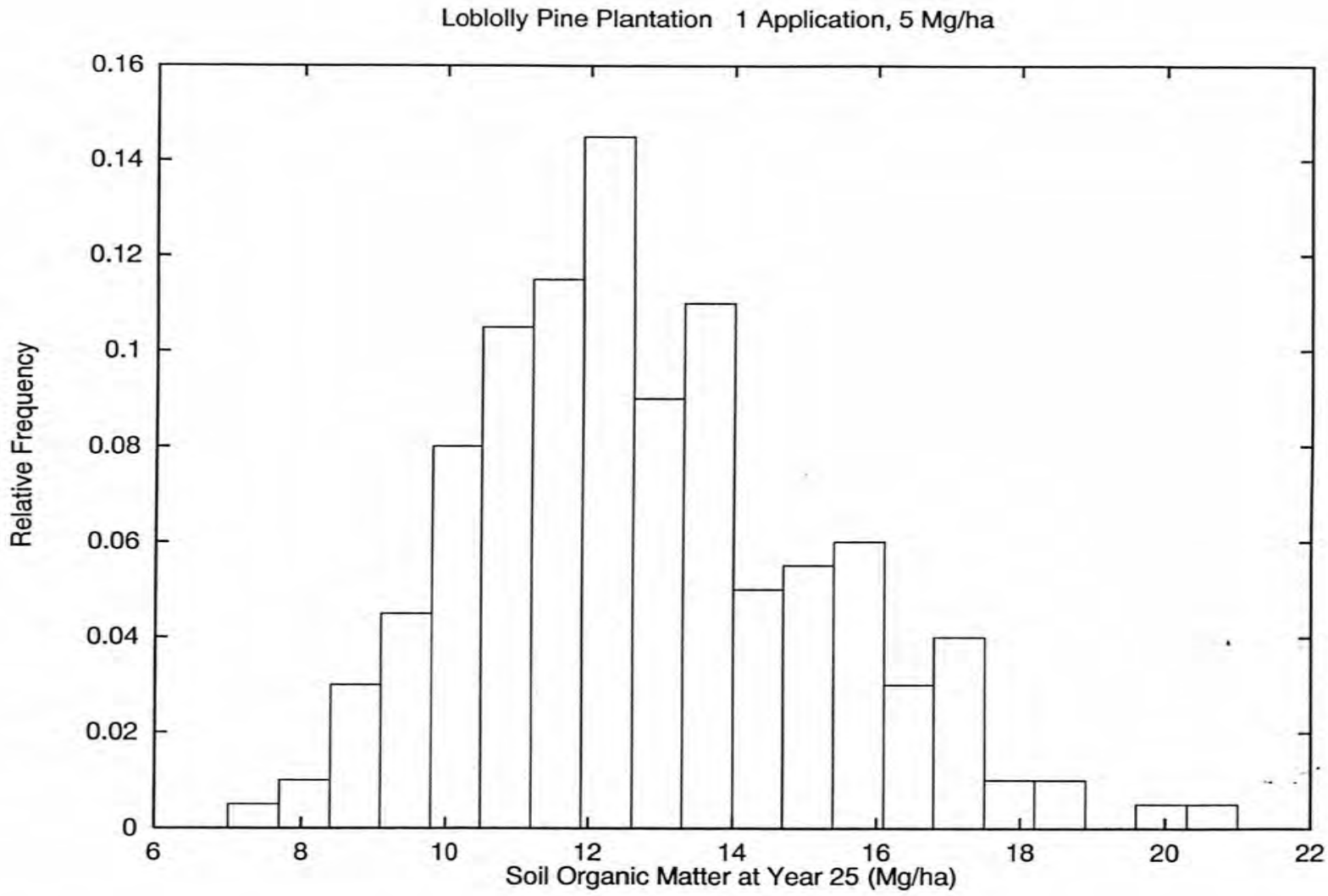

Fig. 2.12a. Frequency distribution of soil organic matter in a loblolly pine plantation at year 25 following a single application of sludge at 5 Mg/ha.

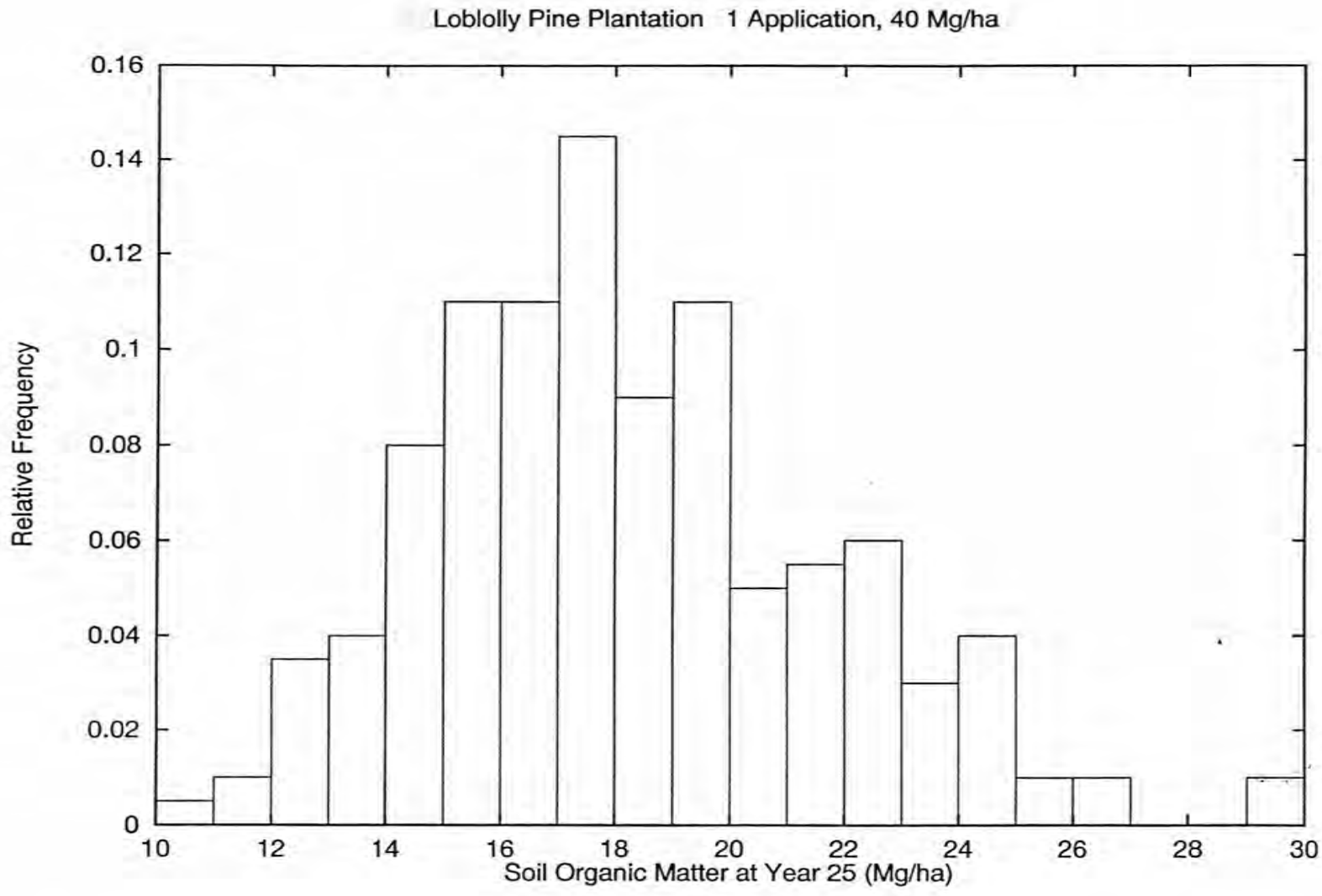

Fig. 2.12b. Frequency distribution of soil organic matter in a loblolly pine plantation at year 25 following a single application of sludge at 40 Mg/ha.

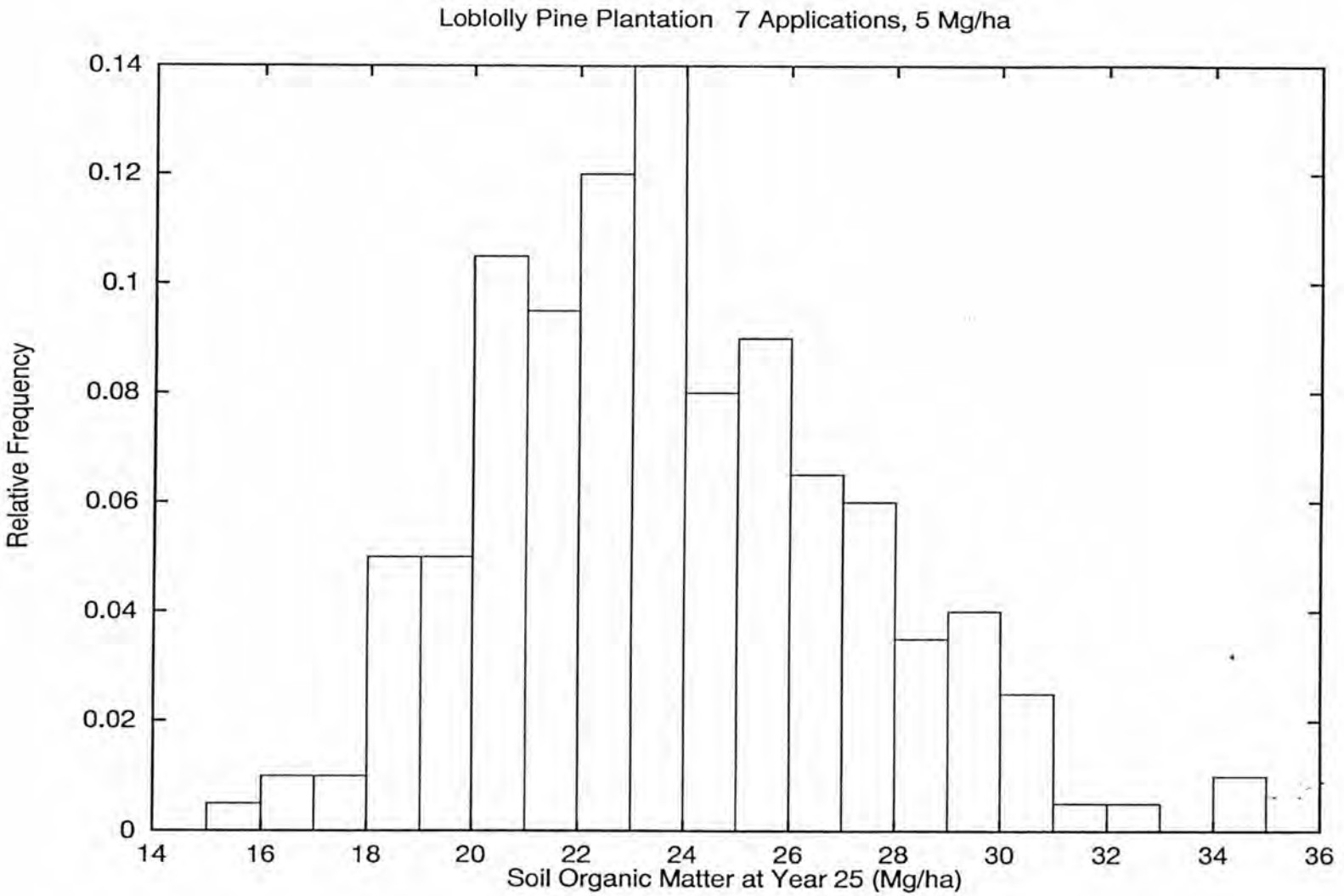

Fig. 2.12c. Frequency distribution of soil organic matter in a loblolly pine plantation at year 25 following seven applications of sludge at 5 Mg/ha, every three years.

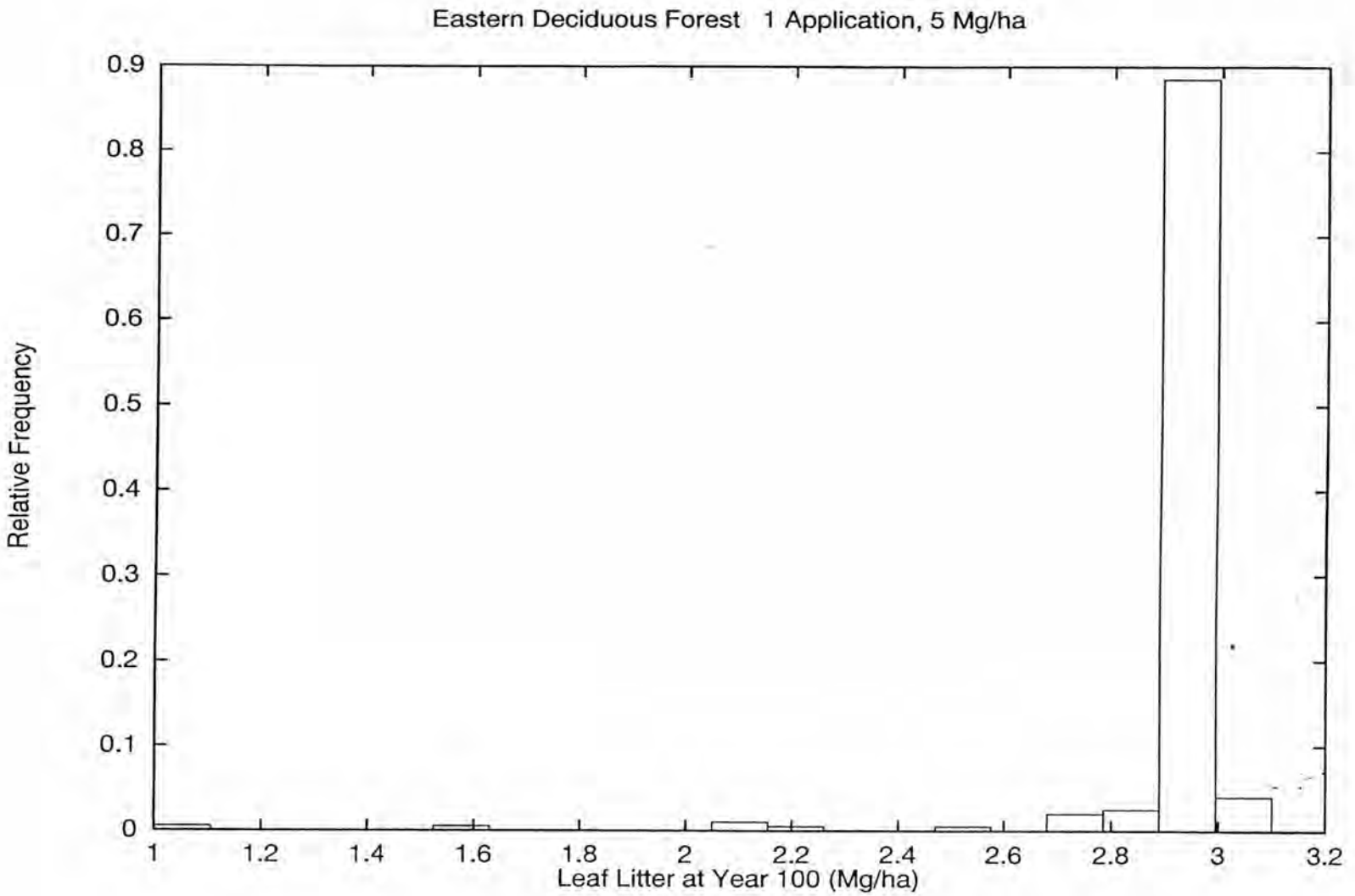

Fig. 2.13a. Frequency distribution of leaf litter in an eastern deciduous forest at year 100 following a single application of sludge at 5 Mg/ha.

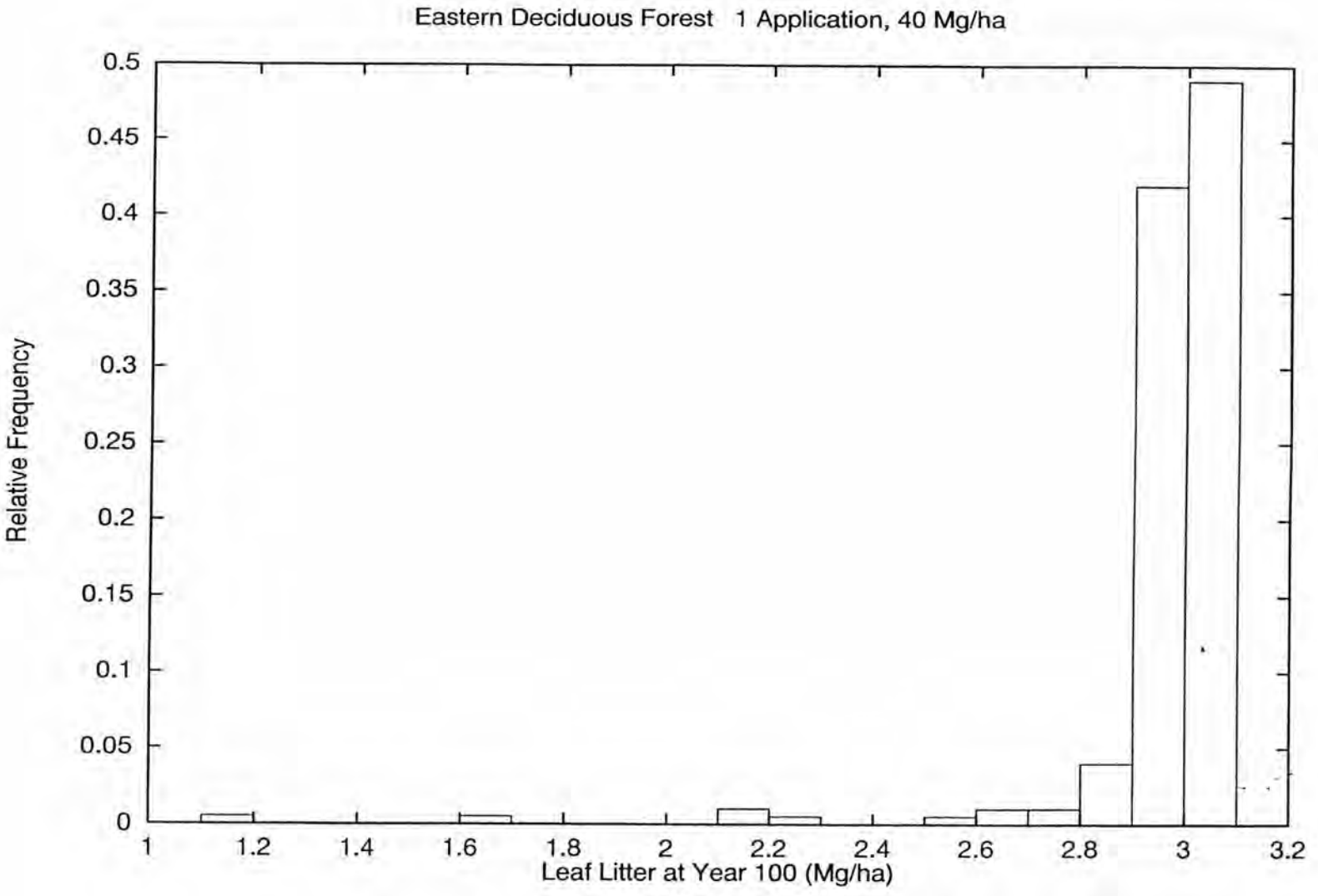

Fig. 2.13b. Frequency distribution of leaf litter in an eastern deciduous forest at year 100 following a single application of sludge at 40 Mg/ha.

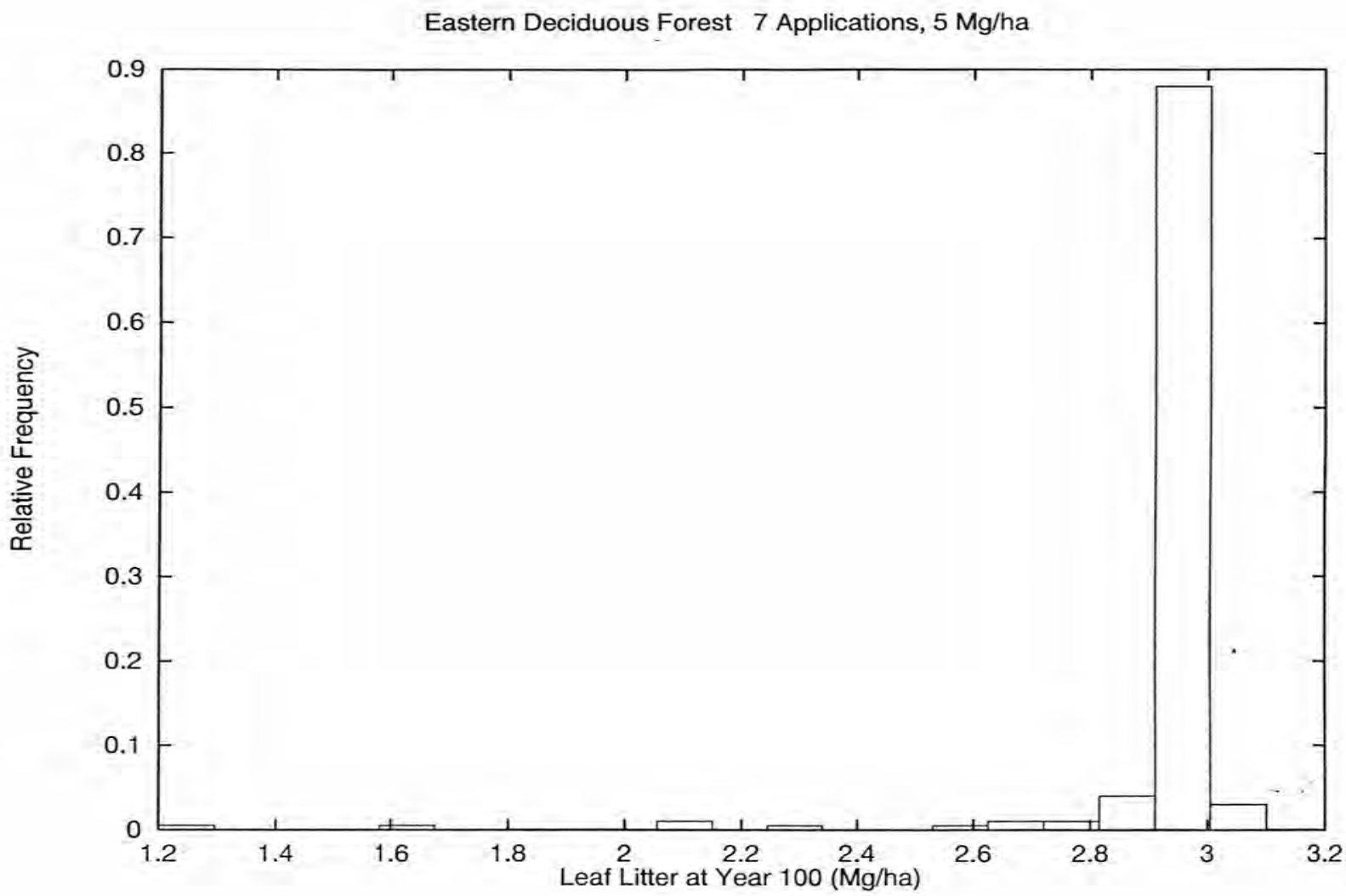

Fig. 2.13c. Frequency distribution of leaf litter in an eastern deciduous forest at year 100 following seven applications of sludge at 5 Mg/ha, every three years.

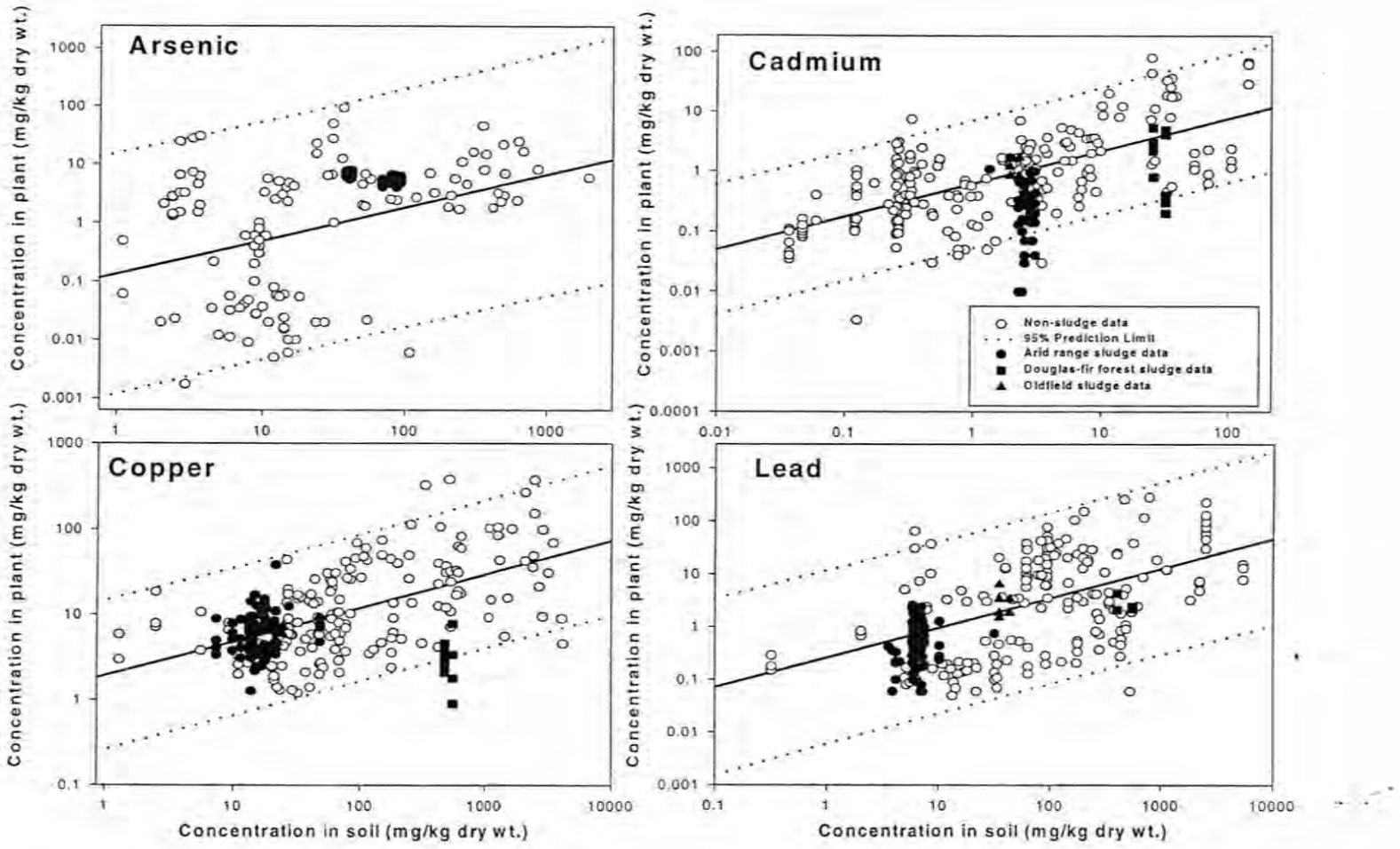

Fig. 4.1. Scatterplots of soil-to-plant foliage bioaccumulation of arsenic, cadmium copper, and lead from non-sludge-amended and sludge-amended soils.

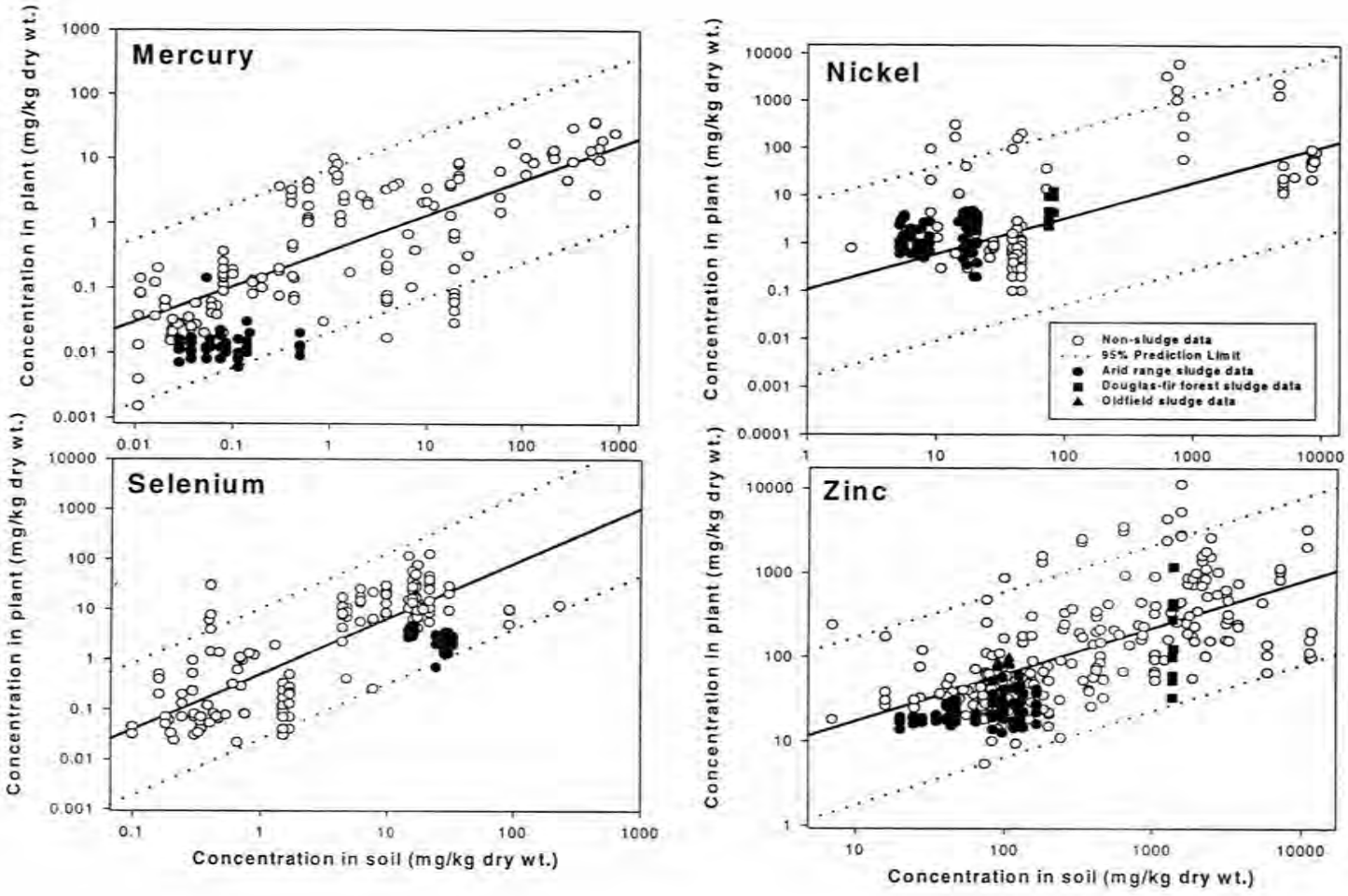

Fig. 4.2. Scatterplots of soil-to-plant foliage bioaccumulation of mercury, nickel, selenium, and zinc from non-sludge-amended and sludge-amended soils.

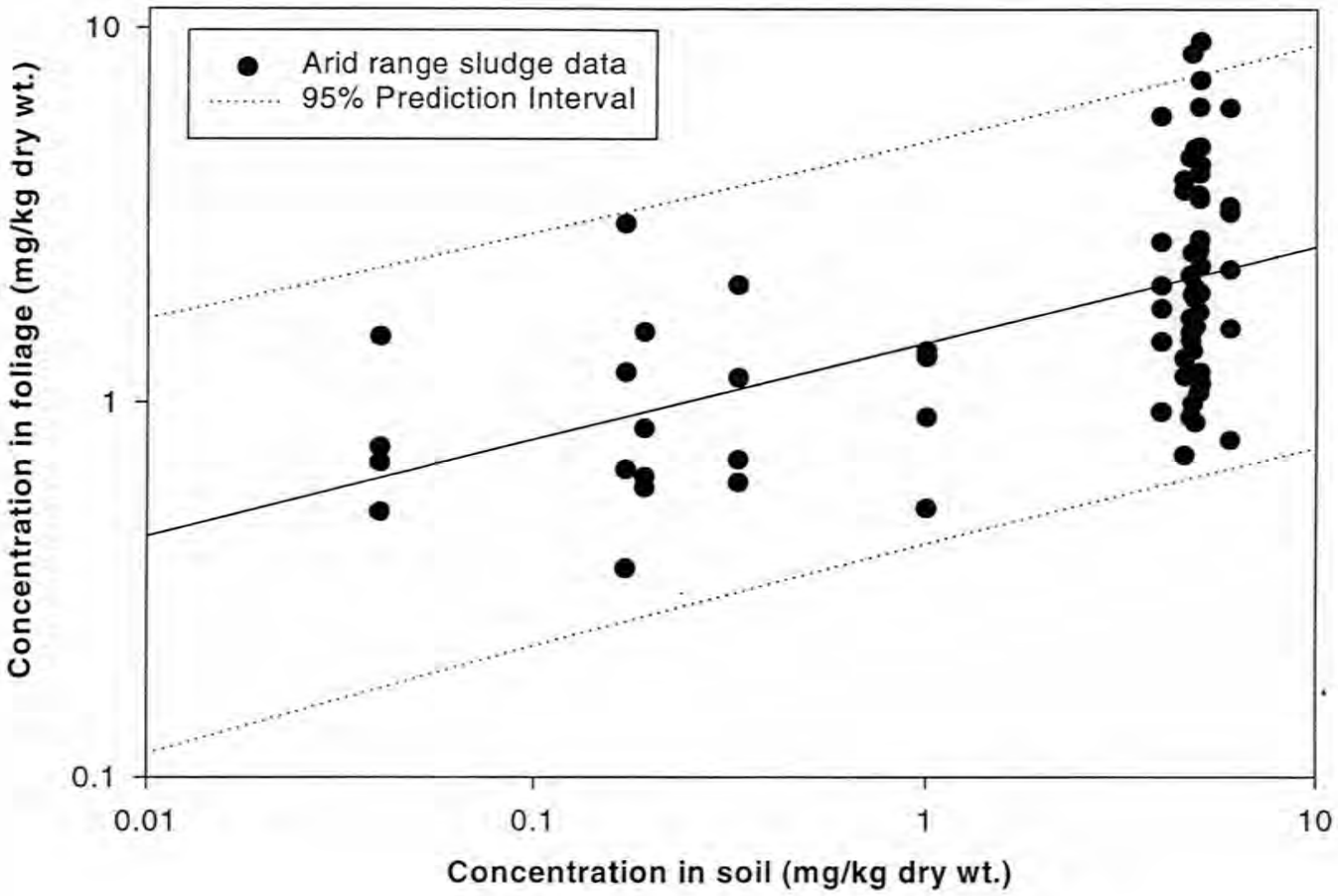

Fig. 4.3. Scatterplot of soil-to-plant foliage bioaccumulation of molybdenum from sludge-amended soils.

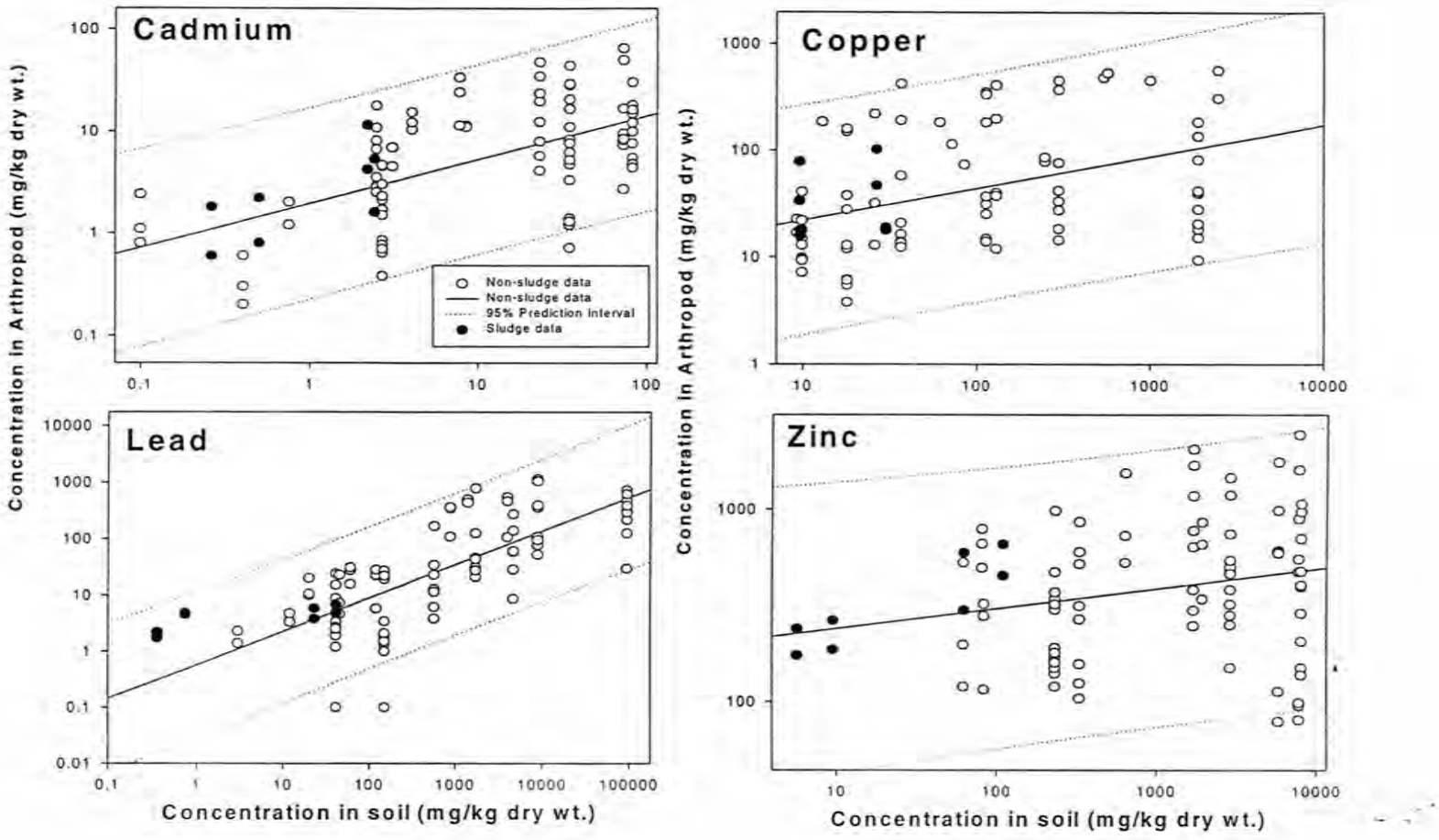

Fig. 4.4. Scatterplots of soil-to-arthropod bioaccumulation of cadmium, copper, lead, and zinc from non-sludge-amended and sludge-amended soils.

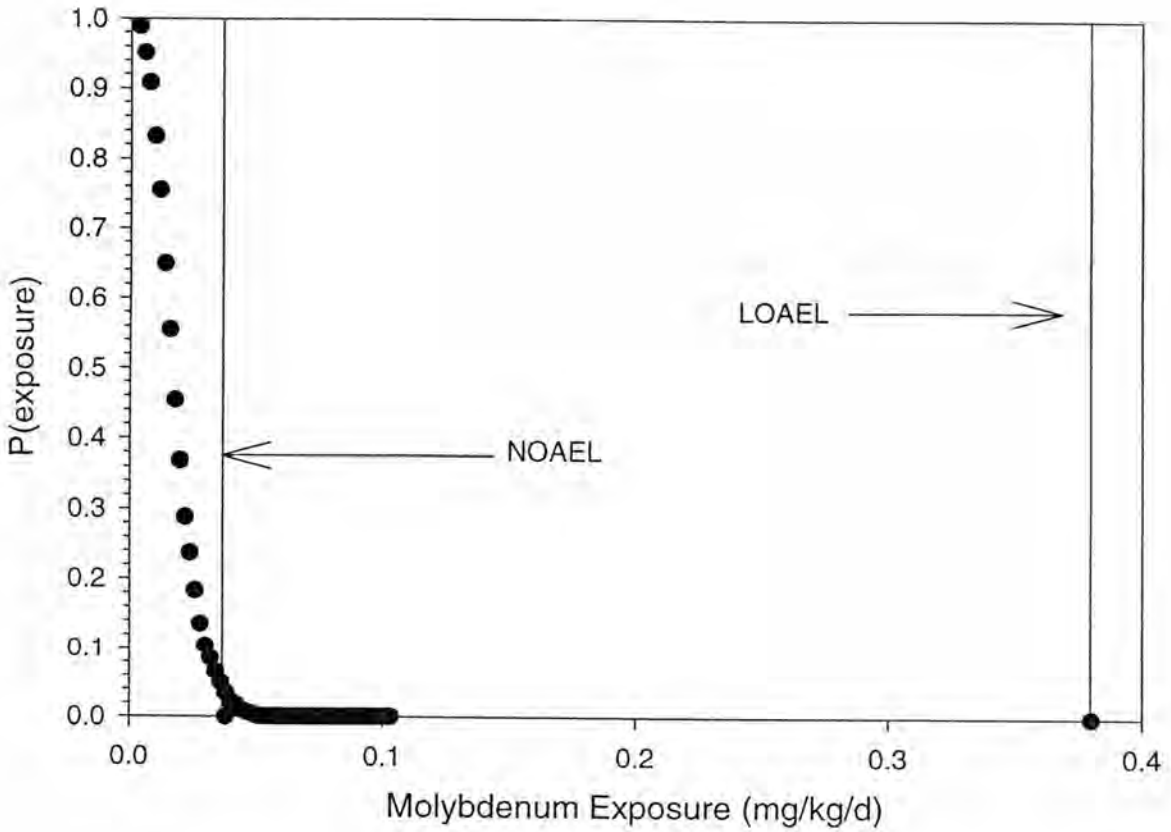

Fig. 4.5. Comparison of estimated distributions for molybdenum exposure of black-tailed deer in the Douglas-fir forest to NOAELs and LOAELs. Monte Carlo methods were used to generate exposure distributions.

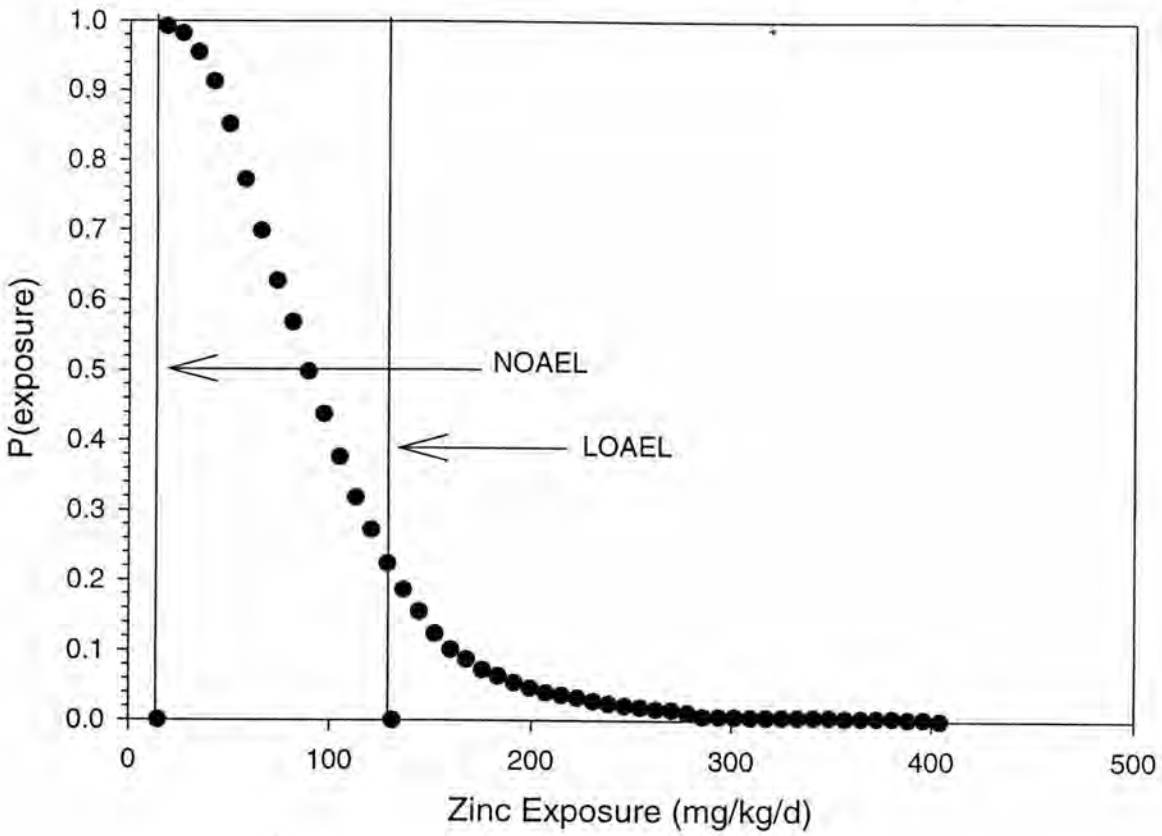

Fig. 4.6. Comparison of estimated distributions for zinc exposure of American robins in the Douglas-fir forest to NOAELs and LOAELs. Monte Carlo methods were used to generate exposure distributions.

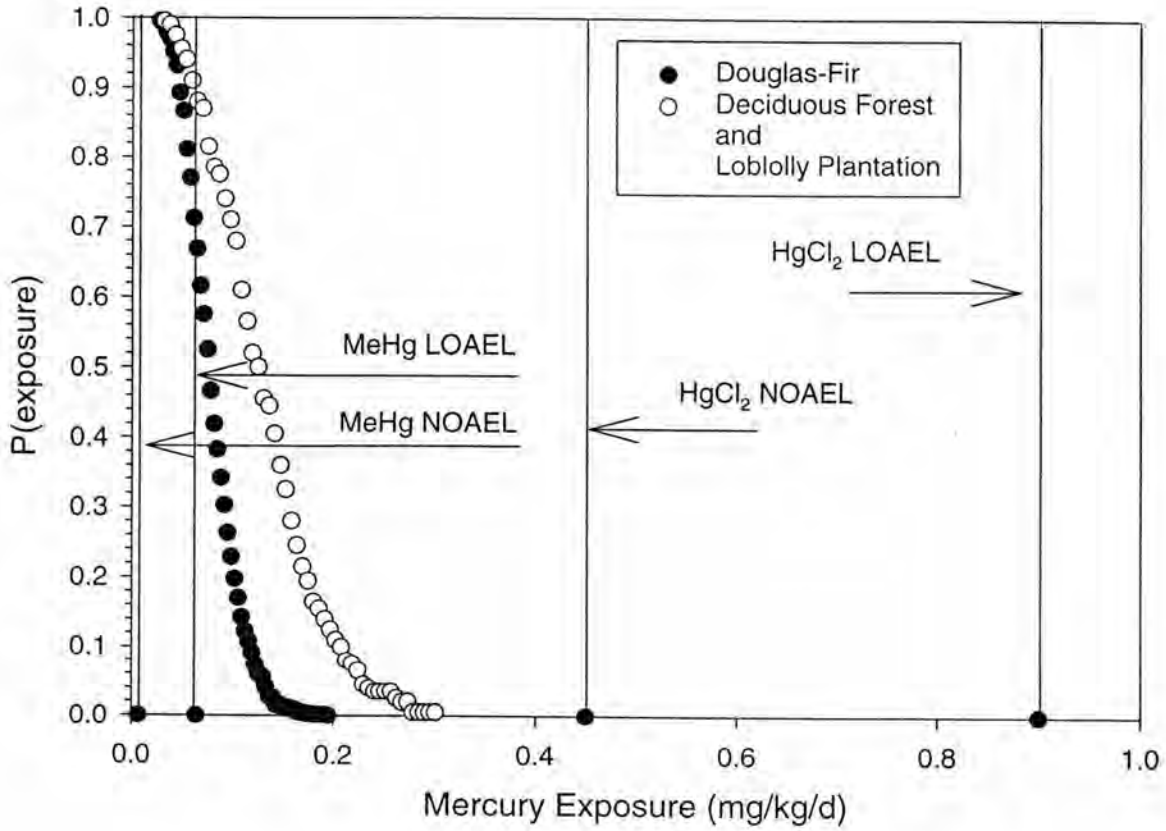

Fig. 4.7. Comparison of estimated distributions for mercury exposure of American robins in the Douglas-fir forest, Eastern deciduous forest, and Loblolly pine plantation to NOAELs and LOAELs methyl mercury (MeHg) and mercuric chloride ($HgCl_2$). Monte Carlo methods were used to generate exposure distributions.

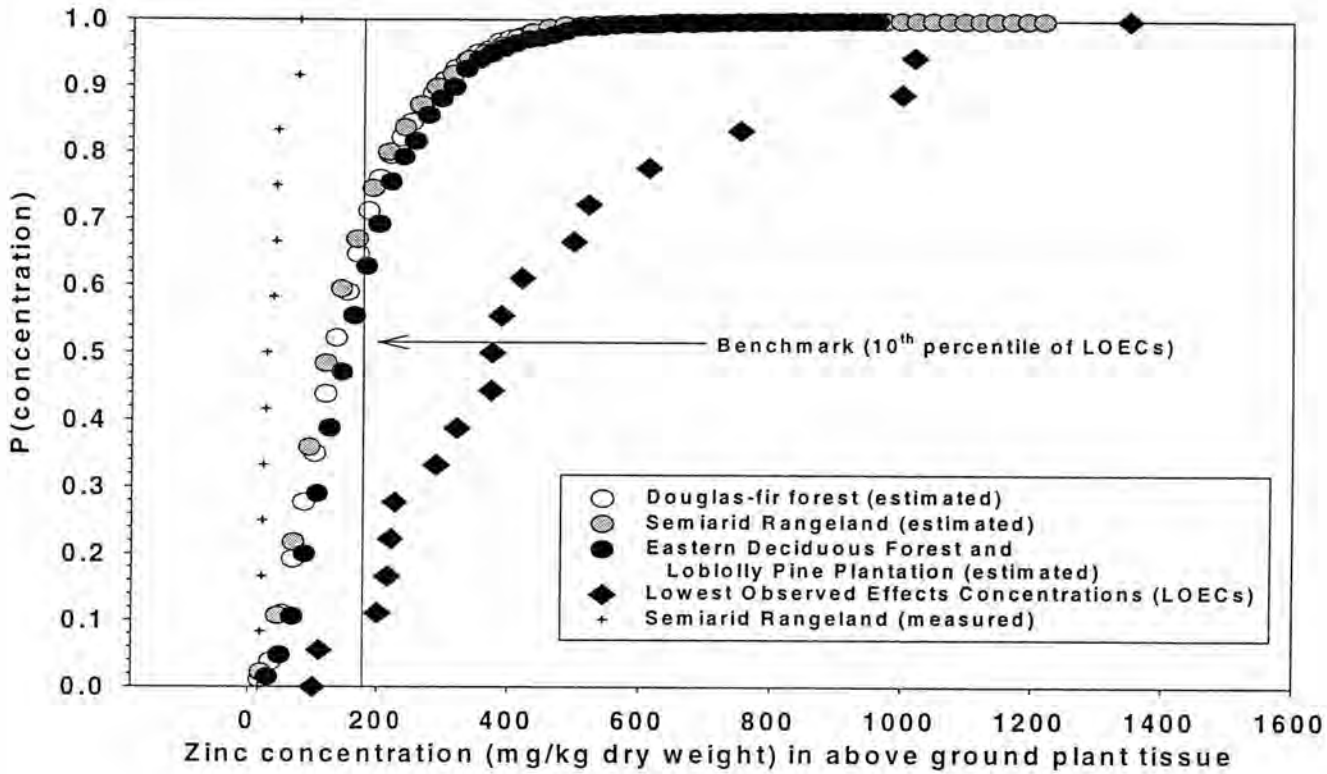

Fig. 4.8. Comparison of estimated distributions for zinc exposure of plants in the four ecosystems (40 Mg/ha application) and measured distributions of exposure in the rangeland ecosystem (45 Mg/ha in Rio Puerco, New Mexico, Fresquez et al. 1991, Fresquez et al. 1990; 40 Mg/ha in Colorado, Harris-Pierce 1994) to Lowest Observed Effects Concentrations in plant tissue.

APPENDIX 1

Concentrations of chemicals in soil and plants at sewage-sludge amended sites

Chemical	Habital Type	Test Family	Test Species	Latin	Time Post-trt	Number Applic.	Applic. Rate (Mg/ha)	Sludge Conc (mg/kg)	Soil Conc (mg/kg)	Soil Extraction	pH	Tissue Type	Tissue Conc (mg/kg)	Uptake Factor	Reference
Arsenic	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	2.5	3.03	78	total	5.6	foliage	5.7	0.073077	Harris-Pierce 1994
Arsenic	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	5	3.03	93.7	total	5.8	foliage	5.7	0.060832	Harris-Pierce 1994
Arsenic	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	10	3.03	87	total	5.5	foliage	4	0.045977	Harris-Pierce 1994
Arsenic	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	21	3.03	80.2	total	5.4	foliage	5.1	0.063591	Harris-Pierce 1994
Arsenic	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	30	3.03	69.2	total	5.6	foliage	5.5	0.07948	Harris-Pierce 1994
Arsenic	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	2.5	3.03	78	total	5.6	foliage	4.7	0.060256	Harris-Pierce 1994
Arsenic	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	5	3.03	93.7	total	5.8	foliage	5.2	0.055496	Harris-Pierce 1994
Arsenic	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	10	3.03	87	total	5.5	foliage	6.3	0.072414	Harris-Pierce 1994
Arsenic	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	21	3.03	80.2	total	5.4	foliage	6.6	0.082294	Harris-Pierce 1994
Arsenic	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	30	3.03	69.2	total	5.6	foliage	4.9	0.070809	Harris-Pierce 1994
Arsenic	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	2.5	3.03	78	total	5.6	foliage	4.9	0.062821	Harris-Pierce 1994
Arsenic	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	5	3.03	93.7	total	5.8	foliage	4.8	0.051227	Harris-Pierce 1994
Arsenic	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	10	3.03	87	total	5.5	foliage	4.2	0.048276	Harris-Pierce 1994
Arsenic	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	21	3.03	80.2	total	5.4	foliage	4.8	0.05985	Harris-Pierce 1994
Arsenic	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	30	3.03	69.2	total	5.6	foliage	4.6	0.066474	Harris-Pierce 1994
Arsenic	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	2.5	3.03	78	total	5.6	foliage	4.7	0.060256	Harris-Pierce 1994
Arsenic	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	5	3.03	93.7	total	5.8	foliage	6.2	0.066169	Harris-Pierce 1994
Arsenic	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	10	3.03	87	total	5.5	foliage	4.3	0.049425	Harris-Pierce 1994
Arsenic	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	21	3.03	80.2	total	5.4	foliage	5.1	0.063591	Harris-Pierce 1994
Arsenic	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	30	3.03	69.2	total	5.6	foliage	4.1	0.059249	Harris-Pierce 1994
Arsenic	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	2.5	3.03	38.1	total	5.5	foliage	6.6	0.173228	Harris-Pierce 1994
Arsenic	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	5	3.03	41.5	total	5.2	foliage	6.7	0.161446	Harris-Pierce 1994
Arsenic	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	10	3.03	40.6	total	5.2	foliage	6.2	0.152709	Harris-Pierce 1994
Arsenic	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	21	3.03	39.8	total	5.3	foliage	6.5	0.163317	Harris-Pierce 1994
Arsenic	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	30	3.03	42	total	5.2	foliage	7.1	0.169048	Harris-Pierce 1994
Arsenic	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	2.5	3.03	38.1	total	5.5	foliage	6.1	0.160105	Harris-Pierce 1994
Arsenic	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	3.03	41.5	total	5.2	foliage	6.3	0.151807	Harris-Pierce 1994
Arsenic	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	3.03	40.6	total	5.2	foliage	6.4	0.157635	Harris-Pierce 1994
Arsenic	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	21	3.03	39.8	total	5.3	foliage	6.4	0.160804	Harris-Pierce 1994
Arsenic	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	3.03	42	total	5.2	foliage	7.7	0.183333	Harris-Pierce 1994
Arsenic	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	2.5	3.03	38.1	total	5.5	foliage	7	0.183727	Harris-Pierce 1994
Arsenic	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	5	3.03	41.5	total	5.2	foliage	5.6	0.13494	Harris-Pierce 1994
Arsenic	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	10	3.03	40.6	total	5.2	foliage	5.5	0.135468	Harris-Pierce 1994
Arsenic	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	21	3.03	39.8	total	5.3	foliage	6.2	0.155779	Harris-Pierce 1994
Arsenic	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	30	3.03	42	total	5.2	foliage	6.5	0.154762	Harris-Pierce 1994
Arsenic	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	2.5	3.03	38.1	total	5.5	foliage	7.6	0.199475	Harris-Pierce 1994
Arsenic	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	5	3.03	41.5	total	5.2	foliage	7.6	0.183133	Harris-Pierce 1994
Arsenic	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	10	3.03	40.6	total	5.2	foliage	7.8	0.192118	Harris-Pierce 1994
Arsenic	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	21	3.03	39.8	total	5.3	foliage	7.4	0.18593	Harris-Pierce 1994
Arsenic	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	30	3.03	42	total	5.2	foliage	7.8	0.185714	Harris-Pierce 1994
Cadmium	Arid range	grass	winter wheat	Triticum aestivum	multi	variable	?		1.35	total	7.1	foliage	1.08	0.8	Baxter et al. 1983
Cadmium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	4.78	2.54	total	7.1	foliage	0.07	0.027559	Pierce 1994
Cadmium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	4.78	2.86	total	7.1	foliage	0.07	0.024476	Pierce 1994
Cadmium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	15	4.78	2.52	total	7	foliage	0.04	0.015873	Pierce 1994
Cadmium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	20	4.78	2.8	total	6.9	foliage	0.14	0.05	Pierce 1994
Cadmium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	25	4.78	2.42	total	7.1	foliage	0.16	0.066116	Pierce 1994
Cadmium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	4.78	3.05	total	6.9	foliage	0.14	0.045902	Pierce 1994
Cadmium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	35	4.78	2.26	total	7.1	foliage	0.13	0.057522	Pierce 1994
Cadmium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	40	4.78	2.9	total	7.1	foliage	0.18	0.062069	Pierce 1994
Cadmium	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	25	4.78	2.42	total	7.1	foliage	0.01	0.004132	Pierce 1994

Cadmium	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	30	4.78	3.05	total	6.9	foliage	0.04	0.013115	Pierce 1994	
Cadmium	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	35	4.78	2.26	total	7.1	foliage	0.01	0.004425	Pierce 1994	
Cadmium	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	40	4.78	2.9	total	7.1	foliage	0.07	0.024138	Pierce 1994	
Cadmium	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	5	4.78	2.54	total	7.1	foliage	0.15	0.059055	Pierce 1994	
Cadmium	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	10	4.78	2.86	total	7.1	foliage	0.16	0.055944	Pierce 1994	
Cadmium	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	15	4.78	2.52	total	7	foliage	0.03	0.011905	Pierce 1994	
Cadmium	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	20	4.78	2.8	total	6.9	foliage	0.2	0.071429	Pierce 1994	
Cadmium	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	25	4.78	2.42	total	7.1	foliage	0.1	0.041322	Pierce 1994	
Cadmium	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	30	4.78	3.05	total	6.9	foliage	0.2	0.065574	Pierce 1994	
Cadmium	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	35	4.78	2.26	total	7.1	foliage	0.13	0.057522	Pierce 1994	
Cadmium	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	40	4.78	2.9	total	7.1	foliage	0.18	0.062069	Pierce 1994	
Cadmium	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	5	4.78	2.54	total	7.1	foliage	0.27	0.106299	Pierce 1994	
Cadmium	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	10	4.78	2.86	total	7.1	foliage	0.32	0.111888	Pierce 1994	
Cadmium	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	15	4.78	2.52	total	7	foliage	0.32	0.126984	Pierce 1994	
Cadmium	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	20	4.78	2.8	total	6.9	foliage	0.28	0.1	Pierce 1994	
Cadmium	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	25	4.78	2.42	total	7.1	foliage	0.32	0.132231	Pierce 1994	
Cadmium	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	30	4.78	3.05	total	6.9	foliage	0.29	0.095082	Pierce 1994	
Cadmium	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	35	4.78	2.26	total	7.1	foliage	0.25	0.110619	Pierce 1994	
Cadmium	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	40	4.78	2.9	total	7.1	foliage	0.21	0.072414	Pierce 1994	
Cadmium	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	5	4.78	2.54	total	7.1	foliage	0.61	0.240157	Pierce 1994	
Cadmium	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	10	4.78	2.86	total	7.1	foliage	0.7	0.244755	Pierce 1994	
Cadmium	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	15	4.78	2.52	total	7	foliage	0.58	0.230159	Pierce 1994	
Cadmium	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	20	4.78	2.8	total	6.9	foliage	0.95	0.339286	Pierce 1994	
Cadmium	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	25	4.78	2.42	total	7.1	foliage	0.66	0.272727	Pierce 1994	
Cadmium	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	30	4.78	3.05	total	6.9	foliage	0.99	0.32459	Pierce 1994	
Cadmium	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	35	4.78	2.26	total	7.1	foliage	0.73	0.323009	Pierce 1994	
Cadmium	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	40	4.78	2.9	total	7.1	foliage	1	0.344828	Pierce 1994	
Cadmium	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	5	4.78	2.54	total	7.1	foliage	0.43	0.169291	Pierce 1994	
Cadmium	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	10	4.78	2.86	total	7.1	foliage	0.33	0.115385	Pierce 1994	
Cadmium	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	15	4.78	2.52	total	7	foliage	0.4	0.15873	Pierce 1994	
Cadmium	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	20	4.78	2.8	total	6.9	foliage	0.32	0.114286	Pierce 1994	
Cadmium	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	25	4.78	2.42	total	7.1	foliage	0.31	0.128099	Pierce 1994	
Cadmium	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	30	4.78	3.05	total	6.9	foliage	0.43	0.140984	Pierce 1994	
Cadmium	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	35	4.78	2.26	total	7.1	foliage	0.33	0.146018	Pierce 1994	
Cadmium	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	40	4.78	2.9	total	7.1	foliage	0.31	0.106897	Pierce 1994	
Cadmium	NW forest	tree	hybrid poplar	Populus trichocarpa x	7 yr; 4 pos	1	512	46.8	32.2	nitric-perchl	4.6	foliage	4.1	0.127329	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	NW forest	tree	lombardy poplar	Populus nigra	7 yr; 4 pos	1	512	46.8	32.2	nitric-perchl	4.6	foliage	5	0.15528	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	NW forest	tree	Douglas-fir	Pseudotsuga menzies	7 yr; 4 pos	1	512	46.8	32.2	nitric-perchl	4.6	foliage	0.2	0.006211	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	NW forest	tree	W. red dedar	Thuja plicata	7 yr; 4 pos	1	512	46.8	32.2	nitric-perchl	4.6	foliage	0.4	0.012422	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	NW forest	tree	W. hemlock	Tsuga heterophylla	7 yr; 4 pos	1	512	46.8	32.2	nitric-perchl	4.6	foliage	0.2	0.006211	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	NW forest	tree	Sitka spruce	Picea sitchensis	7 yr; 4 pos	1	512	46.8	32.2	nitric-perchl	4.6	foliage	0.4	0.012422	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	NW forest	tree	ponderosa pine	Pinus ponderosa	7 yr; 4 pos	1	512	46.8	32.2	nitric-perchl	4.6	foliage	0.3	0.009317	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	NW forest	tree	lombardy poplar	Populus nigra	7 yr; 2-3 p	1	512	46.8	25.6	nitric-perchl	4.7	foliage	5.4	0.210938	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	NW forest	tree	black cottonwood	Populus trichocarpa	7 yr; 2-3 p	1	512	46.8	25.6	nitric-perchl	4.7	foliage	2.2	0.085938	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	NW forest	tree	paper birch	Betula paperifera	7 yr; 2-3 p	1	512	46.8	25.6	nitric-perchl	4.7	foliage	3	0.117188	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	NW forest	tree	Douglas-fir	Pseudotsuga menzies	7 yr; 2-3 p	1	512	46.8	25.6	nitric-perchl	4.7	foliage	0.8	0.03125	Greenleaf-Jenkins and Zasoski 1986	
Cadmium	old field	grass	Japanese brome	Bromus japonicus	10 yr after	50	1.792	19.8-59	1.94	nitric acid, h	4.4	whole st	1.7	0.876289	Levine et al. 1989	
Cadmium	old field	grass	bluegrass	Poa spp.	10 yr after	50	1.792	19.8-59	1.94	nitric acid, h	4.4	whole st	0.85	0.438144	Levine et al. 1989	
Cadmium	old field		woody plant	Rubus frondosis	10 yr after	50	1.792	19.8-59	1.94	nitric acid, h	4.4	1st 10 c	1.2	0.618557	Levine et al. 1989	
Cadmium	old field	grass	foxtail		10 yr after	50	1.792	19.8-59	1.94	nitric acid, h	4.4	whole st	1.68	0.865979	Levine et al. 1989	
Cadmium	old field	grass	blue grass	Poa spp.	about 10 y	50	1.792	19.8	2.28	nitric acid, h	999		0.85	0.372807	Hall et al. 1990	
Cadmium	old field	grass	Japanese brome	Bromus japonicus	about 10 y	50	1.792	19.8	2.28	nitric acid, h	999		1.7	0.745614	Hall et al. 1990	
Copper	Arid range	grass	winter wheat	?	multi	variable	?	Triticum a	27.1	total	7.1	foliage	12.5	0.461255	Baxter et al. 1983	
Copper	Arid range	grass	western wheatgrass	2 yr	1	5	567	Agropyron	14.7	total	7.1	foliage	3.3	0.22449	Pierce 1994	

Copper	Arid range	grass	western wheatgrass	2 yr	1	10	567	Agropyron	17.5	total	7.1	foliage	3.28	0.187429	Pierce 1994
Copper	Arid range	grass	western wheatgrass	2 yr	1	15	567	Agropyron	13.8	total	7	foliage	3.1	0.224638	Pierce 1994
Copper	Arid range	grass	western wheatgrass	2 yr	1	20	567	Agropyron	18	total	6.9	foliage	3.72	0.206667	Pierce 1994
Copper	Arid range	grass	western wheatgrass	2 yr	1	25	567	Agropyron	16.7	total	7.1	foliage	3.74	0.223952	Pierce 1994
Copper	Arid range	grass	western wheatgrass	2 yr	1	30	567	Agropyron	17.5	total	6.9	foliage	3.52	0.201143	Pierce 1994
Copper	Arid range	grass	western wheatgrass	2 yr	1	35	567	Agropyron	15.2	total	7.1	foliage	3.88	0.255263	Pierce 1994
Copper	Arid range	grass	western wheatgrass	2 yr	1	40	567	Agropyron	21.3	total	7.1	foliage	3.99	0.187324	Pierce 1994
Copper	Arid range	grass	bluebunch wheatgras	2 yr	1	5	567	Agropyron	14.7	total	7.1	foliage	3.82	0.259864	Pierce 1994
Copper	Arid range	grass	bluebunch wheatgras	2 yr	1	10	567	Agropyron	17.5	total	7.1	foliage	3.57	0.204	Pierce 1994
Copper	Arid range	grass	bluebunch wheatgras	2 yr	1	15	567	Agropyron	13.8	total	7	foliage	3.96	0.286957	Pierce 1994
Copper	Arid range	grass	bluebunch wheatgras	2 yr	1	20	567	Agropyron	18	total	6.9	foliage	3.93	0.218333	Pierce 1994
Copper	Arid range	grass	bluebunch wheatgras	2 yr	1	25	567	Agropyron	16.7	total	7.1	foliage	3.68	0.220359	Pierce 1994
Copper	Arid range	grass	bluebunch wheatgras	2 yr	1	30	567	Agropyron	17.5	total	6.9	foliage	4.45	0.254286	Pierce 1994
Copper	Arid range	grass	bluebunch wheatgras	2 yr	1	35	567	Agropyron	15.2	total	7.1	foliage	3.93	0.258553	Pierce 1994
Copper	Arid range	grass	bluebunch wheatgras	2 yr	1	40	567	Agropyron	21.3	total	7.1	foliage	4.77	0.223944	Pierce 1994
Copper	Arid range	grass	Indian ricegrass	2 yr	1	5	567	Oryzopsis	14.7	total	7.1	foliage	2.19	0.14898	Pierce 1994
Copper	Arid range	grass	Indian ricegrass	2 yr	1	10	567	Oryzopsis	17.5	total	7.1	foliage	2.82	0.161143	Pierce 1994
Copper	Arid range	grass	Indian ricegrass	2 yr	1	15	567	Oryzopsis	13.8	total	7	foliage	1.27	0.092029	Pierce 1994
Copper	Arid range	grass	Indian ricegrass	2 yr	1	20	567	Oryzopsis	18	total	6.9	foliage	3.08	0.171111	Pierce 1994
Copper	Arid range	grass	Indian ricegrass	2 yr	1	25	567	Oryzopsis	16.7	total	7.1	foliage	2.48	0.148503	Pierce 1994
Copper	Arid range	grass	Indian ricegrass	2 yr	1	30	567	Oryzopsis	17.5	total	6.9	foliage	3.31	0.189143	Pierce 1994
Copper	Arid range	grass	Indian ricegrass	2 yr	1	35	567	Oryzopsis	15.2	total	7.1	foliage	2.48	0.163158	Pierce 1994
Copper	Arid range	grass	Indian ricegrass	2 yr	1	40	567	Oryzopsis	21.3	total	7.1	foliage	3.42	0.160563	Pierce 1994
Copper	Arid range	forb	pulse milkvetch	2 yr	1	5	567	Astragalus	14.7	total	7.1	foliage	6.47	0.440136	Pierce 1994
Copper	Arid range	forb	pulse milkvetch	2 yr	1	10	567	Astragalus	17.5	total	7.1	foliage	6.64	0.379429	Pierce 1994
Copper	Arid range	forb	pulse milkvetch	2 yr	1	15	567	Astragalus	13.8	total	7	foliage	8.56	0.62029	Pierce 1994
Copper	Arid range	forb	pulse milkvetch	2 yr	1	20	567	Astragalus	18	total	6.9	foliage	6.54	0.363333	Pierce 1994
Copper	Arid range	forb	pulse milkvetch	2 yr	1	25	567	Astragalus	16.7	total	7.1	foliage	7.18	0.42994	Pierce 1994
Copper	Arid range	forb	pulse milkvetch	2 yr	1	30	567	Astragalus	17.5	total	6.9	foliage	6.67	0.381143	Pierce 1994
Copper	Arid range	forb	pulse milkvetch	2 yr	1	35	567	Astragalus	15.2	total	7.1	foliage	6.37	0.419079	Pierce 1994
Copper	Arid range	forb	pulse milkvetch	2 yr	1	40	567	Astragalus	21.3	total	7.1	foliage	6.3	0.295775	Pierce 1994
Copper	Arid range	shrub	big sagebrush	2 yr	1	5	567	Artemisia	14.7	total	7.1	foliage	7.67	0.521769	Pierce 1994
Copper	Arid range	shrub	big sagebrush	2 yr	1	10	567	Artemisia	17.5	total	7.1	foliage	7.58	0.433143	Pierce 1994
Copper	Arid range	shrub	big sagebrush	2 yr	1	15	567	Artemisia	13.8	total	7	foliage	8.01	0.580435	Pierce 1994
Copper	Arid range	shrub	big sagebrush	2 yr	1	20	567	Artemisia	18	total	6.9	foliage	8.44	0.468889	Pierce 1994
Copper	Arid range	shrub	big sagebrush	2 yr	1	25	567	Artemisia	16.7	total	7.1	foliage	6.66	0.398802	Pierce 1994
Copper	Arid range	shrub	big sagebrush	2 yr	1	30	567	Artemisia	17.5	total	6.9	foliage	9.36	0.534857	Pierce 1994
Copper	Arid range	shrub	big sagebrush	2 yr	1	35	567	Artemisia	15.2	total	7.1	foliage	6.54	0.430263	Pierce 1994
Copper	Arid range	shrub	big sagebrush	2 yr	1	40	567	Artemisia	21.3	total	7.1	foliage	11	0.516432	Pierce 1994
Copper	Arid range	shrub	Douglas rabbitbrush	2 yr	1	5	567	Chrysothai	14.7	total	7.1	foliage	17	1.156463	Pierce 1994
Copper	Arid range	shrub	Douglas rabbitbrush	2 yr	1	10	567	Chrysothai	17.5	total	7.1	foliage	13.5	0.771429	Pierce 1994
Copper	Arid range	shrub	Douglas rabbitbrush	2 yr	1	15	567	Chrysothai	13.8	total	7	foliage	14.2	1.028986	Pierce 1994
Copper	Arid range	shrub	Douglas rabbitbrush	2 yr	1	20	567	Chrysothai	18	total	6.9	foliage	11.7	0.65	Pierce 1994
Copper	Arid range	shrub	Douglas rabbitbrush	2 yr	1	25	567	Chrysothai	16.7	total	7.1	foliage	10.1	0.60479	Pierce 1994
Copper	Arid range	shrub	Douglas rabbitbrush	2 yr	1	30	567	Chrysothai	17.5	total	6.9	foliage	15.2	0.868571	Pierce 1994
Copper	Arid range	shrub	Douglas rabbitbrush	2 yr	1	35	567	Chrysothai	15.2	total	7.1	foliage	12.7	0.835526	Pierce 1994
Copper	Arid range	shrub	Douglas rabbitbrush	2 yr	1	40	567	Chrysothai	21.3	total	7.1	foliage	38.1	1.788732	Pierce 1994
Copper	Arid range	grass	blue grama	1 yr	1	2.5	553	Bouteloua	11.5	total	5.6	foliage	5.2	0.452174	Harris-Pierce 1994
Copper	Arid range	grass	blue grama	1 yr	1	5	553	Bouteloua	13.4	total	5.8	foliage	7.1	0.529851	Harris-Pierce 1994
Copper	Arid range	grass	blue grama	1 yr	1	10	553	Bouteloua	14.2	total	5.5	foliage	5.4	0.380282	Harris-Pierce 1994
Copper	Arid range	grass	blue grama	1 yr	1	21	553	Bouteloua	13.9	total	5.4	foliage	7.4	0.532374	Harris-Pierce 1994
Copper	Arid range	grass	blue grama	1 yr	1	30	553	Bouteloua	13.7	total	5.6	foliage	8.2	0.59854	Harris-Pierce 1994
Copper	Arid range	grass	western wheatgrass	1 yr	1	2.5	553	Agropyron	11.5	total	5.6	foliage	3.1	0.269565	Harris-Pierce 1994
Copper	Arid range	grass	western wheatgrass	1 yr	1	5	553	Agropyron	13.4	total	5.8	foliage	3.8	0.283582	Harris-Pierce 1994

Copper	Arid range	grass	western wheatgrass	1 yr	1	10	553	Agropyron	14.2	total	5.5	foliage	4.7	0.330986	Harris-Pierce 1994	
Copper	Arid range	grass	western wheatgrass	1 yr	1	21	553	Agropyron	13.9	total	5.4	foliage	4.8	0.345324	Harris-Pierce 1994	
Copper	Arid range	grass	western wheatgrass	1 yr	1	30	553	Agropyron	13.7	total	5.6	foliage	4.4	0.321168	Harris-Pierce 1994	
Copper	Arid range	grass	buffalo grass	1 yr	1	2.5	553	Buchloe d	11.5	total	5.6	foliage	4.1	0.356522	Harris-Pierce 1994	
Copper	Arid range	grass	buffalo grass	1 yr	1	5	553	Buchloe d	13.4	total	5.8	foliage	5.1	0.380597	Harris-Pierce 1994	
Copper	Arid range	grass	buffalo grass	1 yr	1	10	553	Buchloe d	14.2	total	5.5	foliage	5.6	0.394366	Harris-Pierce 1994	
Copper	Arid range	grass	buffalo grass	1 yr	1	21	553	Buchloe d	13.9	total	5.4	foliage	6.4	0.460432	Harris-Pierce 1994	
Copper	Arid range	grass	buffalo grass	1 yr	1	30	553	Buchloe d	13.7	total	5.6	foliage	6.3	0.459854	Harris-Pierce 1994	
Copper	Arid range	shrub	fringed sage	1 yr	1	2.5	553	Artemisia	11.5	total	5.6	foliage	8.8	0.765217	Harris-Pierce 1994	
Copper	Arid range	shrub	fringed sage	1 yr	1	5	553	Artemisia	13.4	total	5.8	foliage	9.2	0.686567	Harris-Pierce 1994	
Copper	Arid range	shrub	fringed sage	1 yr	1	10	553	Artemisia	14.2	total	5.5	foliage	8	0.56338	Harris-Pierce 1994	
Copper	Arid range	shrub	fringed sage	1 yr	1	21	553	Artemisia	13.9	total	5.4	foliage	9.5	0.683453	Harris-Pierce 1994	
Copper	Arid range	shrub	fringed sage	1 yr	1	30	553	Artemisia	13.7	total	5.6	foliage	9.5	0.693431	Harris-Pierce 1994	
Copper	Arid range	grass	blue grama	2 yr	1	2.5	553	Bouteloua	7.4	total	5.5	foliage	4.1	0.554054	Harris-Pierce 1994	
Copper	Arid range	grass	blue grama	2 yr	1	5	553	Bouteloua	9.9	total	5.2	foliage	5.1	0.515152	Harris-Pierce 1994	
Copper	Arid range	grass	blue grama	2 yr	1	10	553	Bouteloua	13.4	total	5.2	foliage	6.3	0.470149	Harris-Pierce 1994	
Copper	Arid range	grass	blue grama	2 yr	1	21	553	Bouteloua	20.8	total	5.3	foliage	6.6	0.317308	Harris-Pierce 1994	
Copper	Arid range	grass	blue grama	2 yr	1	30	553	Bouteloua	48.2	total	5.2	foliage	7.1	0.147303	Harris-Pierce 1994	
Copper	Arid range	grass	western wheatgrass	2 yr	1	2.5	553	Agropyron	7.4	total	5.5	foliage	3.4	0.459459	Harris-Pierce 1994	
Copper	Arid range	grass	western wheatgrass	2 yr	1	5	553	Agropyron	9.9	total	5.2	foliage	3.8	0.383838	Harris-Pierce 1994	
Copper	Arid range	grass	western wheatgrass	2 yr	1	10	553	Agropyron	13.4	total	5.2	foliage	3.9	0.291045	Harris-Pierce 1994	
Copper	Arid range	grass	western wheatgrass	2 yr	1	21	553	Agropyron	20.8	total	5.3	foliage	4.5	0.216346	Harris-Pierce 1994	
Copper	Arid range	grass	western wheatgrass	2 yr	1	30	553	Agropyron	48.2	total	5.2	foliage	4.8	0.099585	Harris-Pierce 1994	
Copper	Arid range	grass	buffalo grass	2 yr	1	2.5	553	Buchloe d	7.4	total	5.5	foliage	5	0.675676	Harris-Pierce 1994	
Copper	Arid range	grass	buffalo grass	2 yr	1	5	553	Buchloe d	9.9	total	5.2	foliage	6.1	0.616162	Harris-Pierce 1994	
Copper	Arid range	grass	buffalo grass	2 yr	1	10	553	Buchloe d	13.4	total	5.2	foliage	6.8	0.507463	Harris-Pierce 1994	
Copper	Arid range	grass	buffalo grass	2 yr	1	21	553	Buchloe d	20.8	total	5.3	foliage	8.4	0.403846	Harris-Pierce 1994	
Copper	Arid range	grass	buffalo grass	2 yr	1	30	553	Buchloe d	48.2	total	5.2	foliage	9.1	0.188797	Harris-Pierce 1994	
Copper	Arid range	shrub	fringed sage	2 yr	1	2.5	553	Artemisia	7.4	total	5.5	foliage	8.9	1.202703	Harris-Pierce 1994	
Copper	Arid range	shrub	fringed sage	2 yr	1	5	553	Artemisia	9.9	total	5.2	foliage	7.9	0.79798	Harris-Pierce 1994	
Copper	Arid range	shrub	fringed sage	2 yr	1	10	553	Artemisia	13.4	total	5.2	foliage	7.6	0.567164	Harris-Pierce 1994	
Copper	Arid range	shrub	fringed sage	2 yr	1	21	553	Artemisia	20.8	total	5.3	foliage	7.6	0.365385	Harris-Pierce 1994	
Copper	Arid range	shrub	fringed sage	2 yr	1	30	553	Artemisia	48.2	total	5.2	foliage	7.7	0.159751	Harris-Pierce 1994	
Copper	NW forest	tree	hybrid poplar	7 yr; 4 post-planting	1	512	699	Populus tri	476.2	nitric-perchl	4.6	foliage	4.4	0.00924	Greenleaf-Jenkins and Zasoski 1986	
Copper	NW forest	tree	lombardy poplar	7 yr; 4 post-planting	1	512	699	Populus ni	476.2	nitric-perchl	4.6	foliage	2.3	0.00483	Greenleaf-Jenkins and Zasoski 1986	
Copper	NW forest	tree	Douglas-fir	7 yr; 4 post-planting	1	512	699	Pseudotsu	476.2	nitric-perchl	4.6	foliage	4.1	0.00861	Greenleaf-Jenkins and Zasoski 1986	
Copper	NW forest	tree	W. red dedar	7 yr; 4 post-planting	1	512	699	Thuja plica	476.2	nitric-perchl	4.6	foliage	4.6	0.00966	Greenleaf-Jenkins and Zasoski 1986	
Copper	NW forest	tree	W. hemlock	7 yr; 4 post-planting	1	512	699	Tsuga het	476.2	nitric-perchl	4.6	foliage	2.9	0.00609	Greenleaf-Jenkins and Zasoski 1986	
Copper	NW forest	tree	Sitka spruce	7 yr; 4 post-planting	1	512	699	Picea sitch	476.2	nitric-perchl	4.6	foliage	3.3	0.00693	Greenleaf-Jenkins and Zasoski 1986	
Copper	NW forest	tree	ponderosa pine	7 yr; 4 post-planting	1	512	699	Pinus ponc	476.2	nitric-perchl	4.6	foliage	2.1	0.00441	Greenleaf-Jenkins and Zasoski 1986	
Copper	NW forest	tree	lombardy poplar	7 yr; 2-3 post-plantir	1	512	699	Populus ni	547.7	nitric-perchl	4.7	foliage	1.8	0.003286	Greenleaf-Jenkins and Zasoski 1986	
Copper	NW forest	tree	black cottonwood	7 yr; 2-3 post-plantir	1	512	699	Populus tri	547.7	nitric-perchl	4.7	foliage	0.9	0.001643	Greenleaf-Jenkins and Zasoski 1986	
Copper	NW forest	tree	paper birch	7 yr; 2-3 post-plantir	1	512	699	Betula pap	547.7	nitric-perchl	4.7	foliage	7.8	0.014241	Greenleaf-Jenkins and Zasoski 1986	
Copper	NW forest	tree	Douglas-fir	7 yr; 2-3 post-plantir	1	512	699	Pseudotsu	547.7	nitric-perchl	4.7	foliage	3.4	0.006208	Greenleaf-Jenkins and Zasoski 1986	
Copper	old field	grass	Japanese brome	10 yr after 1st tmt;	50	1.792	320-380.6	Bromus jap	22	nitric acid	4.4	whole st	5.92	0.269091	Levine et al. 1989	
Copper	old field	grass	bluegrass	10 yr after 1st tmt;	50	1.792	320-380.6	Poa spp.	22	nitric acid	4.4	whole st	7.37	0.335	Levine et al. 1989	
Copper	old field		woody plant	10 yr after 1st tmt;	50	1.792	320-380.6	Rubus fron	22	nitric acid	4.4	1st 10 c	4.7	0.213636	Levine et al. 1989	
Copper	old field	grass	foxtail	10 yr after 1st tmt;	50	1.792	320-380.6		22	nitric acid	4.4	whole st	6.34	0.288182	Levine et al. 1989	
Copper	old field	grass	blue grass	Poa spp.	about 10 y	50	1.792	Poa spp.	24.19	nitric acid, h	999		7.37	0.304671	Hall et al. 1990	
Copper	old field	grass	Japanese brome	Bromus japonicus	about 10 y	50	1.792	Bromus jap	24.19	nitric acid, h	999		5.92	0.244729	Hall et al. 1990	
Lead	Arid range	grass	winter wheat	Triticum aestivum	?	multi	variable	?	31.8	total	7.1	foliage	0.75	0.023585	Baxter et al. 1983	
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	47.4	5.87	total	7.1	foliage	0.29	0.049404	Pierce 1994	
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	47.4	7.03	total	7.1	foliage	0.37	0.052632	Pierce 1994	
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	15	47.4	5.87	total	7	foliage	0.21	0.035775	Pierce 1994	

Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	20	47.4	6.94	total	6.9	foliage	0.43	0.06196	Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	25	47.4	5.99	total	7.1	foliage	0.39	0.065109	Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	47.4	6.14	total	6.9	foliage	0.33	0.053746	Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	35	47.4	5.98	total	7.1	foliage	0.1	0.016722	Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	40	47.4	7.26	total	7.1	foliage	0.27	0.03719	Pierce 1994
Lead	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	5	47.4	5.87	total	7.1	foliage	0.5	0.085179	Pierce 1994
Lead	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	10	47.4	7.03	total	7.1	foliage	0.06	0.008535	Pierce 1994
Lead	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	35	47.4	5.98	total	7.1	foliage	0.2	0.033445	Pierce 1994
Lead	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	40	47.4	7.26	total	7.1	foliage	0.06	0.008264	Pierce 1994
Lead	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	5	47.4	5.87	total	7.1	foliage	2.25	0.383305	Pierce 1994
Lead	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	10	47.4	7.03	total	7.1	foliage	2.34	0.332859	Pierce 1994
Lead	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	15	47.4	5.87	total	7	foliage	2.56	0.436116	Pierce 1994
Lead	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	20	47.4	6.94	total	6.9	foliage	1.7	0.244957	Pierce 1994
Lead	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	25	47.4	5.99	total	7.1	foliage	2.32	0.387312	Pierce 1994
Lead	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	30	47.4	6.14	total	6.9	foliage	2.13	0.346906	Pierce 1994
Lead	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	35	47.4	5.98	total	7.1	foliage	1.76	0.294314	Pierce 1994
Lead	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	40	47.4	7.26	total	7.1	foliage	1.3	0.179063	Pierce 1994
Lead	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	5	47.4	5.87	total	7.1	foliage	0.66	0.112436	Pierce 1994
Lead	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	10	47.4	7.03	total	7.1	foliage	0.58	0.082504	Pierce 1994
Lead	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	15	47.4	5.87	total	7	foliage	0.54	0.091993	Pierce 1994
Lead	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	20	47.4	6.94	total	6.9	foliage	0.81	0.116715	Pierce 1994
Lead	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	25	47.4	5.99	total	7.1	foliage	0.54	0.09015	Pierce 1994
Lead	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	30	47.4	6.14	total	6.9	foliage	0.36	0.058632	Pierce 1994
Lead	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	35	47.4	5.98	total	7.1	foliage	0.56	0.093645	Pierce 1994
Lead	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	40	47.4	7.26	total	7.1	foliage	0.6	0.082645	Pierce 1994
Lead	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	5	47.4	5.87	total	7.1	foliage	1.18	0.201022	Pierce 1994
Lead	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	10	47.4	7.03	total	7.1	foliage	1.22	0.173542	Pierce 1994
Lead	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	15	47.4	5.87	total	7	foliage	1.21	0.206133	Pierce 1994
Lead	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	20	47.4	6.94	total	6.9	foliage	0.85	0.122478	Pierce 1994
Lead	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	25	47.4	5.99	total	7.1	foliage	0.91	0.15192	Pierce 1994
Lead	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	30	47.4	6.14	total	6.9	foliage	1.22	0.198697	Pierce 1994
Lead	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	35	47.4	5.98	total	7.1	foliage	0.73	0.122074	Pierce 1994
Lead	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	40	47.4	7.26	total	7.1	foliage	0.85	0.11708	Pierce 1994
Lead	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	2.5	117	3.9	total	5.6	foliage	0.06	0.015385	Harris-Pierce 1994
Lead	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	5	117	4.2	total	5.8	foliage	0.21	0.05	Harris-Pierce 1994
Lead	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	21	117	4.2	total	5.4	foliage	0.1	0.02381	Harris-Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	2.5	117	3.9	total	5.6	foliage	0.36	0.092308	Harris-Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	5	117	4.2	total	5.8	foliage	0.33	0.078571	Harris-Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	10	117	4.5	total	5.5	foliage	0.22	0.048889	Harris-Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	21	117	4.2	total	5.4	foliage	0.22	0.052381	Harris-Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	30	117	3.6	total	5.6	foliage	0.42	0.116667	Harris-Pierce 1994
Lead	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	2.5	117	6	total	5.5	foliage	0.92	0.153333	Harris-Pierce 1994
Lead	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	5	117	6.5	total	5.2	foliage	0.96	0.147692	Harris-Pierce 1994
Lead	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	10	117	7.1	total	5.2	foliage	1.24	0.174648	Harris-Pierce 1994
Lead	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	21	117	7.5	total	5.3	foliage	1	0.133333	Harris-Pierce 1994
Lead	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	30	117	10.3	total	5.2	foliage	1.28	0.124272	Harris-Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	2.5	117	6	total	5.5	foliage	0.69	0.115	Harris-Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	117	6.5	total	5.2	foliage	0.36	0.055385	Harris-Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	117	7.1	total	5.2	foliage	0.44	0.061972	Harris-Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	21	117	7.5	total	5.3	foliage	0.5	0.066667	Harris-Pierce 1994
Lead	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	117	10.3	total	5.2	foliage	0.44	0.042718	Harris-Pierce 1994
Lead	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	2.5	117	6	total	5.5	foliage	0.13	0.021667	Harris-Pierce 1994
Lead	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	5	117	6.5	total	5.2	foliage	0.16	0.024615	Harris-Pierce 1994
Lead	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	10	117	7.1	total	5.2	foliage	0.08	0.011268	Harris-Pierce 1994

Lead	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	21	117	7.5	total	5.3	foliage	0.67	0.089333	Harris-Pierce 1994
Lead	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	30	117	10.3	total	5.2	foliage	0.24	0.023301	Harris-Pierce 1994
Lead	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	5	117	6.5	total	5.2	foliage	0.09	0.013846	Harris-Pierce 1994
Lead	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	10	117	7.1	total	5.2	foliage	0.26	0.03662	Harris-Pierce 1994
Lead	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	21	117	7.5	total	5.3	foliage	0.19	0.025333	Harris-Pierce 1994
Lead	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	30	117	10.3	total	5.2	foliage	0.28	0.027184	Harris-Pierce 1994
Lead	NW forest	tree	hybrid poplar	Populus trichocarpa x	7 yr; 4 pos	1	512	1210	548.5	nitric-perchl	4.6	foliage	2.2	0.004011	Greenleaf-Jenkins and Zasoski 1986
Lead	NW forest	tree	lombardy poplar	Populus nigra	7 yr; 4 pos	1	512	1210	548.5	nitric-perchl	4.6	foliage	2.4	0.004376	Greenleaf-Jenkins and Zasoski 1986
Lead	NW forest	tree	black cottonwood	Populus trichocarpa	7 yr; 2-3 p	1	512	1210	401.6	nitric-perchl	4.7	foliage	2.1	0.005229	Greenleaf-Jenkins and Zasoski 1986
Lead	NW forest	tree	paper birch	Betula paperifera	7 yr; 2-3 p	1	512	1210	401.6	nitric-perchl	4.7	foliage	4.3	0.010707	Greenleaf-Jenkins and Zasoski 1986
Lead	old field	grass	Japanese brome	Bromus japonicus	10 yr after	50	1.792	243-473	35	nitric acid	4.4	whole st	1.85	0.052857	Levine et al. 1989
Lead	old field	grass	bluegrass	Poa spp.	10 yr after	50	1.792	243-473	35	nitric acid	4.4	whole st	3.38	0.096571	Levine et al. 1989
Lead	old field		woody plant	Rubus frondosis	10 yr after	50	1.792	243-473	35	nitric acid	4.4	1st 10 c	1.5	0.042857	Levine et al. 1989
Lead	old field	grass	foxtail		10 yr after	50	1.792	243-473	35	nitric acid	4.4	whole st	6.39	0.182571	Levine et al. 1989
Lead	old field	grass	blue grass	Poa spp.	about 10 y	50	1.792	19.8	43.48	nitric acid, l	999		3.38	0.077737	Hall et al. 1990
Lead	old field	grass	Japanese brome	Bromus japonicus	about 10 y	50	1.792	19.8	43.48	nitric acid, l	999		1.85	0.042548	Hall et al. 1990
Mercury	Arid range	grass	winter wheat	Triticum aestivum	?	multi	variable	?	0.15	total	7.1	foliage	0.02	0.133333	Baxter et al. 1983
Mercury	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	2.5	6.06	0.028	total	5.6	foliage	0.013	0.464286	Harris-Pierce 1994
Mercury	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	5	6.06	0.037	total	5.8	foliage	0.01	0.27027	Harris-Pierce 1994
Mercury	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	10	6.06	0.058	total	5.5	foliage	0.016	0.275862	Harris-Pierce 1994
Mercury	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	21	6.06	0.037	total	5.4	foliage	0.014	0.378378	Harris-Pierce 1994
Mercury	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	30	6.06	0.075	total	5.6	foliage	0.008	0.106667	Harris-Pierce 1994
Mercury	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	2.5	6.06	0.028	total	5.6	foliage	0.016	0.571429	Harris-Pierce 1994
Mercury	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	5	6.06	0.037	total	5.8	foliage	0.015	0.405405	Harris-Pierce 1994
Mercury	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	10	6.06	0.058	total	5.5	foliage	0.016	0.275862	Harris-Pierce 1994
Mercury	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	21	6.06	0.037	total	5.4	foliage	0.016	0.432432	Harris-Pierce 1994
Mercury	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	30	6.06	0.075	total	5.6	foliage	0.022	0.293333	Harris-Pierce 1994
Mercury	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	2.5	6.06	0.028	total	5.6	foliage	0.007	0.25	Harris-Pierce 1994
Mercury	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	5	6.06	0.037	total	5.8	foliage	0.017	0.459459	Harris-Pierce 1994
Mercury	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	10	6.06	0.058	total	5.5	foliage	0.013	0.224138	Harris-Pierce 1994
Mercury	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	21	6.06	0.037	total	5.4	foliage	0.016	0.432432	Harris-Pierce 1994
Mercury	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	30	6.06	0.075	total	5.6	foliage	0.018	0.24	Harris-Pierce 1994
Mercury	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	2.5	6.06	0.028	total	5.6	foliage	0.011	0.392857	Harris-Pierce 1994
Mercury	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	5	6.06	0.037	total	5.8	foliage	0.011	0.297297	Harris-Pierce 1994
Mercury	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	10	6.06	0.058	total	5.5	foliage	0.012	0.206897	Harris-Pierce 1994
Mercury	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	21	6.06	0.037	total	5.4	foliage	0.008	0.216216	Harris-Pierce 1994
Mercury	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	30	6.06	0.075	total	5.6	foliage	0.012	0.16	Harris-Pierce 1994
Mercury	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	2.5	6.06	0.053	total	5.5	foliage	0.008	0.150943	Harris-Pierce 1994
Mercury	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	5	6.06	0.084	total	5.2	foliage	0.011	0.130952	Harris-Pierce 1994
Mercury	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	10	6.06	0.114	total	5.2	foliage	0.006	0.052632	Harris-Pierce 1994
Mercury	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	21	6.06	0.14	total	5.3	foliage	0.012	0.085714	Harris-Pierce 1994
Mercury	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	30	6.06	0.493	total	5.2	foliage	0.013	0.026369	Harris-Pierce 1994
Mercury	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	2.5	6.06	0.053	total	5.5	foliage	0.14	2.641509	Harris-Pierce 1994
Mercury	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	6.06	0.084	total	5.2	foliage	0.01	0.119048	Harris-Pierce 1994
Mercury	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	6.06	0.114	total	5.2	foliage	0.008	0.070175	Harris-Pierce 1994
Mercury	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	21	6.06	0.14	total	5.3	foliage	0.01	0.071429	Harris-Pierce 1994
Mercury	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	6.06	0.493	total	5.2	foliage	0.009	0.018256	Harris-Pierce 1994
Mercury	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	2.5	6.06	0.053	total	5.5	foliage	0.011	0.207547	Harris-Pierce 1994
Mercury	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	5	6.06	0.084	total	5.2	foliage	0.014	0.166667	Harris-Pierce 1994
Mercury	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	10	6.06	0.114	total	5.2	foliage	0.01	0.087719	Harris-Pierce 1994
Mercury	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	21	6.06	0.14	total	5.3	foliage	0.03	0.214286	Harris-Pierce 1994
Mercury	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	30	6.06	0.493	total	5.2	foliage	0.02	0.040568	Harris-Pierce 1994
Mercury	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	2.5	6.06	0.053	total	5.5	foliage	0.012	0.226415	Harris-Pierce 1994
Mercury	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	5	6.06	0.084	total	5.2	foliage	0.012	0.142857	Harris-Pierce 1994

Mercury	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	10	6.06	0.114	total	5.2	foliage	0.016	0.140351	Harris-Pierce 1994
Mercury	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	21	6.06	0.14	total	5.3	foliage	0.015	0.107143	Harris-Pierce 1994
Mercury	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	30	6.06	0.493	total	5.2	foliage	0.012	0.024341	Harris-Pierce 1994
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	2.92	5.03	total	7.1	foliage	4.07	0.809145	Pierce 1994
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	2.92	6	total	7.1	foliage	2.26	0.376667	Pierce 1994
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	15	2.92	4.82	total	7	foliage	2.51	0.520747	Pierce 1994
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	20	2.92	4.99	total	6.9	foliage	2.6	0.521042	Pierce 1994
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	25	2.92	4.01	total	7.1	foliage	1.45	0.361596	Pierce 1994
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	2.92	4.76	total	6.9	foliage	2.19	0.460084	Pierce 1994
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	35	2.92	4.89	total	7.1	foliage	1.89	0.386503	Pierce 1994
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	40	2.92	4.6	total	7.1	foliage	1.3	0.282609	Pierce 1994
Molybdenu	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	5	2.92	5.03	total	7.1	foliage	3.49	0.693837	Pierce 1994
Molybdenu	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	10	2.92	6	total	7.1	foliage	3.21	0.535	Pierce 1994
Molybdenu	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	15	2.92	4.82	total	7	foliage	1.64	0.340249	Pierce 1994
Molybdenu	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	20	2.92	4.99	total	6.9	foliage	1.74	0.348697	Pierce 1994
Molybdenu	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	25	2.92	4.01	total	7.1	foliage	2.05	0.511222	Pierce 1994
Molybdenu	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	30	2.92	4.76	total	6.9	foliage	1.54	0.323529	Pierce 1994
Molybdenu	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	35	2.92	4.89	total	7.1	foliage	1.99	0.406953	Pierce 1994
Molybdenu	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	40	2.92	4.6	total	7.1	foliage	1.31	0.284783	Pierce 1994
Molybdenu	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	5	2.92	5.03	total	7.1	foliage	2.72	0.540755	Pierce 1994
Molybdenu	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	10	2.92	6	total	7.1	foliage	1.57	0.261667	Pierce 1994
Molybdenu	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	15	2.92	4.82	total	7	foliage	1.37	0.284232	Pierce 1994
Molybdenu	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	20	2.92	4.99	total	6.9	foliage	2.54	0.509018	Pierce 1994
Molybdenu	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	25	2.92	4.01	total	7.1	foliage	2.68	0.668329	Pierce 1994
Molybdenu	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	30	2.92	4.76	total	6.9	foliage	1.68	0.352941	Pierce 1994
Molybdenu	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	35	2.92	4.89	total	7.1	foliage	2.02	0.413088	Pierce 1994
Molybdenu	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	40	2.92	4.6	total	7.1	foliage	3.67	0.797826	Pierce 1994
Molybdenu	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	5	2.92	5.03	total	7.1	foliage	6.16	1.224652	Pierce 1994
Molybdenu	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	10	2.92	6	total	7.1	foliage	6.12	1.02	Pierce 1994
Molybdenu	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	15	2.92	4.82	total	7	foliage	8.57	1.778008	Pierce 1994
Molybdenu	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	20	2.92	4.99	total	6.9	foliage	3.58	0.717435	Pierce 1994
Molybdenu	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	25	2.92	4.01	total	7.1	foliage	5.82	1.451372	Pierce 1994
Molybdenu	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	30	2.92	4.76	total	6.9	foliage	4.5	0.945378	Pierce 1994
Molybdenu	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	35	2.92	4.89	total	7.1	foliage	4.76	0.973415	Pierce 1994
Molybdenu	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	40	2.92	4.6	total	7.1	foliage	3.93	0.854348	Pierce 1994
Molybdenu	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	5	2.92	5.03	total	7.1	foliage	1.2	0.238569	Pierce 1994
Molybdenu	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	10	2.92	6	total	7.1	foliage	0.79	0.131667	Pierce 1994
Molybdenu	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	15	2.92	4.82	total	7	foliage	0.99	0.205394	Pierce 1994
Molybdenu	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	20	2.92	4.99	total	6.9	foliage	1.06	0.212425	Pierce 1994
Molybdenu	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	25	2.92	4.01	total	7.1	foliage	0.94	0.234414	Pierce 1994
Molybdenu	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	30	2.92	4.76	total	6.9	foliage	0.91	0.191176	Pierce 1994
Molybdenu	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	35	2.92	4.89	total	7.1	foliage	0.88	0.179959	Pierce 1994
Molybdenu	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	40	2.92	4.6	total	7.1	foliage	0.72	0.156522	Pierce 1994
Molybdenu	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	5	2.92	5.03	total	7.1	foliage	1.96	0.389662	Pierce 1994
Molybdenu	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	10	2.92	6	total	7.1	foliage	3.33	0.555	Pierce 1994
Molybdenu	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	15	2.92	4.82	total	7	foliage	1.94	0.40249	Pierce 1994
Molybdenu	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	20	2.92	4.99	total	6.9	foliage	1.14	0.228457	Pierce 1994
Molybdenu	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	25	2.92	4.01	total	7.1	foliage	1.78	0.44389	Pierce 1994
Molybdenu	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	30	2.92	4.76	total	6.9	foliage	1.46	0.306723	Pierce 1994
Molybdenu	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	35	2.92	4.89	total	7.1	foliage	1.59	0.325153	Pierce 1994
Molybdenu	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	40	2.92	4.6	total	7.1	foliage	1.17	0.254348	Pierce 1994
Molybdenu	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	5	16.1	0.33	total	5.8	foliage	1.16	3.515152	Harris-Pierce 1994
Molybdenu	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	30	16.1	0.17	total	5.6	foliage	1.2	7.058824	Harris-Pierce 1994
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	5	16.1	0.33	total	5.8	foliage	0.61	1.848485	Harris-Pierce 1994

Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	30	16.1	0.17	total	5.6	foliage	0.36	2.117647	Harris-Pierce 1994	
Molybdenu	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	5	16.1	0.33	total	5.8	foliage	0.7	2.121212	Harris-Pierce 1994	
Molybdenu	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	30	16.1	0.17	total	5.6	foliage	0.66	3.882353	Harris-Pierce 1994	
Molybdenu	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	5	16.1	0.33	total	5.8	foliage	2.06	6.242424	Harris-Pierce 1994	
Molybdenu	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	30	16.1	0.17	total	5.6	foliage	3	17.64706	Harris-Pierce 1994	
Molybdenu	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	10	16.1	0.04	total	5.2	foliage	0.76	19	Harris-Pierce 1994	
Molybdenu	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	21	16.1	0.19	total	5.3	foliage	0.85	4.473684	Harris-Pierce 1994	
Molybdenu	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	30	16.1	1	total	5.2	foliage	0.91	0.91	Harris-Pierce 1994	
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	16.1	0.04	total	5.2	foliage	0.69	17.25	Harris-Pierce 1994	
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	21	16.1	0.19	total	5.3	foliage	0.63	3.315789	Harris-Pierce 1994	
Molybdenu	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	16.1	1	total	5.2	foliage	1.37	1.37	Harris-Pierce 1994	
Molybdenu	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	10	16.1	0.04	total	5.2	foliage	0.51	12.75	Harris-Pierce 1994	
Molybdenu	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	21	16.1	0.19	total	5.3	foliage	0.59	3.105263	Harris-Pierce 1994	
Molybdenu	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	30	16.1	1	total	5.2	foliage	0.52	0.52	Harris-Pierce 1994	
Molybdenu	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	10	16.1	0.04	total	5.2	foliage	1.5	37.5	Harris-Pierce 1994	
Molybdenu	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	21	16.1	0.19	total	5.3	foliage	1.54	8.105263	Harris-Pierce 1994	
Molybdenu	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	30	16.1	1	total	5.2	foliage	1.32	1.32	Harris-Pierce 1994	
Nickel	Arid range	grass	winter wheat	Triticum aestivum	?	multi	variable	?	15	total	7.1	foliage	2.8	0.186667	Baxter et al. 1983	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	14.7	18.1	total	7.1	foliage	0.64	0.035359	Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	14.7	20.4	total	7.1	foliage	0.97	0.047549	Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	15	14.7	16.2	total	7	foliage	0.79	0.048765	Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	20	14.7	20.8	total	6.9	foliage	1.04	0.05	Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	25	14.7	18.5	total	7.1	foliage	0.91	0.049189	Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	14.7	19.3	total	6.9	foliage	0.94	0.048705	Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	35	14.7	17.2	total	7.1	foliage	0.75	0.043605	Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	40	14.7	19.7	total	7.1	foliage	0.94	0.047716	Pierce 1994	
Nickel	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	5	14.7	18.1	total	7.1	foliage	1.03	0.056906	Pierce 1994	
Nickel	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	10	14.7	20.4	total	7.1	foliage	0.2	0.009804	Pierce 1994	
Nickel	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	15	14.7	16.2	total	7	foliage	0.47	0.029012	Pierce 1994	
Nickel	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	20	14.7	20.8	total	6.9	foliage	0.39	0.01875	Pierce 1994	
Nickel	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	25	14.7	18.5	total	7.1	foliage	0.65	0.035135	Pierce 1994	
Nickel	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	30	14.7	19.3	total	6.9	foliage	0.63	0.032642	Pierce 1994	
Nickel	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	35	14.7	17.2	total	7.1	foliage	0.35	0.020349	Pierce 1994	
Nickel	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	40	14.7	19.7	total	7.1	foliage	0.2	0.010152	Pierce 1994	
Nickel	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	5	14.7	18.1	total	7.1	foliage	1.68	0.092818	Pierce 1994	
Nickel	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	10	14.7	20.4	total	7.1	foliage	1.85	0.090686	Pierce 1994	
Nickel	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	15	14.7	16.2	total	7	foliage	1.24	0.076543	Pierce 1994	
Nickel	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	20	14.7	20.8	total	6.9	foliage	2.56	0.123077	Pierce 1994	
Nickel	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	25	14.7	18.5	total	7.1	foliage	1.89	0.102162	Pierce 1994	
Nickel	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	30	14.7	19.3	total	6.9	foliage	4.78	0.247668	Pierce 1994	
Nickel	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	35	14.7	17.2	total	7.1	foliage	2.03	0.118023	Pierce 1994	
Nickel	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	40	14.7	19.7	total	7.1	foliage	2.71	0.137563	Pierce 1994	
Nickel	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	5	14.7	18.1	total	7.1	foliage	3.93	0.217127	Pierce 1994	
Nickel	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	10	14.7	20.4	total	7.1	foliage	3.77	0.184804	Pierce 1994	
Nickel	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	15	14.7	16.2	total	7	foliage	4.37	0.269753	Pierce 1994	
Nickel	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	20	14.7	20.8	total	6.9	foliage	3.47	0.166827	Pierce 1994	
Nickel	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	25	14.7	18.5	total	7.1	foliage	3.81	0.205946	Pierce 1994	
Nickel	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	30	14.7	19.3	total	6.9	foliage	2.93	0.151813	Pierce 1994	
Nickel	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	35	14.7	17.2	total	7.1	foliage	3.82	0.222093	Pierce 1994	
Nickel	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	40	14.7	19.7	total	7.1	foliage	3.53	0.179188	Pierce 1994	
Nickel	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	5	14.7	18.1	total	7.1	foliage	2.56	0.141436	Pierce 1994	
Nickel	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	10	14.7	20.4	total	7.1	foliage	2.69	0.131863	Pierce 1994	
Nickel	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	15	14.7	16.2	total	7	foliage	4.01	0.247531	Pierce 1994	
Nickel	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	20	14.7	20.8	total	6.9	foliage	3.96	0.190385	Pierce 1994	

Nickel	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	25	14.7	18.5	total	7.1	foliage	3.87	0.209189	Pierce 1994	
Nickel	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	30	14.7	19.3	total	6.9	foliage	3.59	0.18601	Pierce 1994	
Nickel	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	35	14.7	17.2	total	7.1	foliage	4.2	0.244186	Pierce 1994	
Nickel	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	40	14.7	19.7	total	7.1	foliage	4.58	0.232487	Pierce 1994	
Nickel	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	5	14.7	18.1	total	7.1	foliage	3.91	0.216022	Pierce 1994	
Nickel	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	10	14.7	20.4	total	7.1	foliage	3.73	0.182843	Pierce 1994	
Nickel	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	15	14.7	16.2	total	7	foliage	3.85	0.237654	Pierce 1994	
Nickel	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	20	14.7	20.8	total	6.9	foliage	3.02	0.145192	Pierce 1994	
Nickel	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	25	14.7	18.5	total	7.1	foliage	3.76	0.203243	Pierce 1994	
Nickel	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	30	14.7	19.3	total	6.9	foliage	3.86	0.2	Pierce 1994	
Nickel	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	35	14.7	17.2	total	7.1	foliage	4.67	0.271512	Pierce 1994	
Nickel	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscid	2 yr	1	40	14.7	19.7	total	7.1	foliage	3.49	0.177157	Pierce 1994	
Nickel	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	2.5	19.2	8	total	5.6	foliage	1	0.125	Harris-Pierce 1994	
Nickel	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	5	19.2	8.8	total	5.8	foliage	1.4	0.159091	Harris-Pierce 1994	
Nickel	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	10	19.2	7.6	total	5.5	foliage	0.9	0.118421	Harris-Pierce 1994	
Nickel	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	21	19.2	7.1	total	5.4	foliage	1	0.140845	Harris-Pierce 1994	
Nickel	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	30	19.2	6.3	total	5.6	foliage	1.3	0.206349	Harris-Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	2.5	19.2	8	total	5.6	foliage	0.5	0.0625	Harris-Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	5	19.2	8.8	total	5.8	foliage	0.7	0.079545	Harris-Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	10	19.2	7.6	total	5.5	foliage	0.6	0.078947	Harris-Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	21	19.2	7.1	total	5.4	foliage	0.6	0.084507	Harris-Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	30	19.2	6.3	total	5.6	foliage	0.6	0.095238	Harris-Pierce 1994	
Nickel	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	2.5	19.2	8	total	5.6	foliage	0.7	0.0875	Harris-Pierce 1994	
Nickel	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	5	19.2	8.8	total	5.8	foliage	1	0.113636	Harris-Pierce 1994	
Nickel	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	10	19.2	7.6	total	5.5	foliage	0.7	0.092105	Harris-Pierce 1994	
Nickel	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	21	19.2	7.1	total	5.4	foliage	0.8	0.112676	Harris-Pierce 1994	
Nickel	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	30	19.2	6.3	total	5.6	foliage	0.6	0.095238	Harris-Pierce 1994	
Nickel	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	2.5	19.2	8	total	5.6	foliage	2.7	0.3375	Harris-Pierce 1994	
Nickel	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	5	19.2	8.8	total	5.8	foliage	2.8	0.318182	Harris-Pierce 1994	
Nickel	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	10	19.2	7.6	total	5.5	foliage	2	0.263158	Harris-Pierce 1994	
Nickel	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	21	19.2	7.1	total	5.4	foliage	1.9	0.267606	Harris-Pierce 1994	
Nickel	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	30	19.2	6.3	total	5.6	foliage	2	0.31746	Harris-Pierce 1994	
Nickel	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	2.5	19.2	5.2	total	5.5	foliage	1.1	0.211538	Harris-Pierce 1994	
Nickel	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	5	19.2	5.3	total	5.2	foliage	1.1	0.207547	Harris-Pierce 1994	
Nickel	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	10	19.2	5.6	total	5.2	foliage	1.2	0.214286	Harris-Pierce 1994	
Nickel	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	21	19.2	5.5	total	5.3	foliage	1.4	0.254545	Harris-Pierce 1994	
Nickel	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	30	19.2	5.6	total	5.2	foliage	1.5	0.267857	Harris-Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	2.5	19.2	5.2	total	5.5	foliage	0.6	0.115385	Harris-Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	19.2	5.3	total	5.2	foliage	0.7	0.132075	Harris-Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	19.2	5.6	total	5.2	foliage	0.7	0.125	Harris-Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	21	19.2	5.5	total	5.3	foliage	0.9	0.163636	Harris-Pierce 1994	
Nickel	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	19.2	5.6	total	5.2	foliage	0.7	0.125	Harris-Pierce 1994	
Nickel	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	2.5	19.2	5.2	total	5.5	foliage	0.8	0.153846	Harris-Pierce 1994	
Nickel	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	5	19.2	5.3	total	5.2	foliage	1	0.188679	Harris-Pierce 1994	
Nickel	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	10	19.2	5.6	total	5.2	foliage	1.2	0.214286	Harris-Pierce 1994	
Nickel	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	21	19.2	5.5	total	5.3	foliage	1.2	0.218182	Harris-Pierce 1994	
Nickel	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	30	19.2	5.6	total	5.2	foliage	1.2	0.214286	Harris-Pierce 1994	
Nickel	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	2.5	19.2	5.2	total	5.5	foliage	2.4	0.461538	Harris-Pierce 1994	
Nickel	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	5	19.2	5.3	total	5.2	foliage	3	0.566038	Harris-Pierce 1994	
Nickel	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	10	19.2	5.6	total	5.2	foliage	3.4	0.607143	Harris-Pierce 1994	
Nickel	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	21	19.2	5.5	total	5.3	foliage	3.4	0.618182	Harris-Pierce 1994	
Nickel	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	30	19.2	5.6	total	5.2	foliage	3.8	0.678571	Harris-Pierce 1994	
Nickel	NW forest	tree	hybrid poplar	Populus trichocarpa x	7 yr; 4 pos	1	512	123	74.7	nitric-perchl	4.6	foliage	4.7	0.062918	Greenleaf-Jenkins and Zasoski 1986	
Nickel	NW forest	tree	lombardy poplar	Populus nigra	7 yr; 4 pos	1	512	123	74.7	nitric-perchl	4.6	foliage	4.6	0.06158	Greenleaf-Jenkins and Zasoski 1986	

Nickel	NW forest	tree	Douglas-fir	Pseudotsuga menzies	7 yr; 4 pos	1	512	123	74.7	nitric-perchl	4.6	foliage	3.9	0.052209	Greenleaf-Jenkins and Zasoski 1986
Nickel	NW forest	tree	W. red dedar	Thuja plicata	7 yr; 4 pos	1	512	123	74.7	nitric-perchl	4.6	foliage	9.7	0.129853	Greenleaf-Jenkins and Zasoski 1986
Nickel	NW forest	tree	W. hemlock	Tsuga heterophylla	7 yr; 4 pos	1	512	123	74.7	nitric-perchl	4.6	foliage	3.1	0.041499	Greenleaf-Jenkins and Zasoski 1986
Nickel	NW forest	tree	Sitka spruce	Picea sitchensis	7 yr; 4 pos	1	512	123	74.7	nitric-perchl	4.6	foliage	4.2	0.056225	Greenleaf-Jenkins and Zasoski 1986
Nickel	NW forest	tree	ponderosa pine	Pinus ponderosa	7 yr; 4 pos	1	512	123	74.7	nitric-perchl	4.6	foliage	2.3	0.03079	Greenleaf-Jenkins and Zasoski 1986
Nickel	NW forest	tree	lombardy poplar	Populus nigra	7 yr; 2-3 p	1	512	123	81.1	nitric-perchl	4.7	foliage	9.6	0.118372	Greenleaf-Jenkins and Zasoski 1986
Nickel	NW forest	tree	black cottonwood	Populus trichocarpa	7 yr; 2-3 p	1	512	123	81.1	nitric-perchl	4.7	foliage	4.4	0.054254	Greenleaf-Jenkins and Zasoski 1986
Nickel	NW forest	tree	paper birch	Betula paperifera	7 yr; 2-3 p	1	512	123	81.1	nitric-perchl	4.7	foliage	11.4	0.140567	Greenleaf-Jenkins and Zasoski 1986
Nickel	NW forest	tree	Douglas-fir	Pseudotsuga menzies	7 yr; 2-3 p	1	512	123	81.1	nitric-perchl	4.7	foliage	12.3	0.151665	Greenleaf-Jenkins and Zasoski 1986
Selenium	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	2.5	13.6	28.4	total	5.6	foliage	2.7	0.09507	Harris-Pierce 1994
Selenium	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	5	13.6	33.6	total	5.8	foliage	2.9	0.08631	Harris-Pierce 1994
Selenium	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	10	13.6	30.8	total	5.5	foliage	2.1	0.068182	Harris-Pierce 1994
Selenium	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	21	13.6	29.1	total	5.4	foliage	2.5	0.085911	Harris-Pierce 1994
Selenium	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	30	13.6	24.4	total	5.6	foliage	2.9	0.118852	Harris-Pierce 1994
Selenium	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	2.5	13.6	28.4	total	5.6	foliage	2.6	0.091549	Harris-Pierce 1994
Selenium	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	5	13.6	33.6	total	5.8	foliage	2.7	0.080357	Harris-Pierce 1994
Selenium	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	10	13.6	30.8	total	5.5	foliage	3.5	0.113636	Harris-Pierce 1994
Selenium	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	21	13.6	29.1	total	5.4	foliage	3.5	0.120275	Harris-Pierce 1994
Selenium	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	30	13.6	24.4	total	5.6	foliage	3.1	0.127049	Harris-Pierce 1994
Selenium	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	2.5	13.6	28.4	total	5.6	foliage	1.5	0.052817	Harris-Pierce 1994
Selenium	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	5	13.6	33.6	total	5.8	foliage	1.9	0.056548	Harris-Pierce 1994
Selenium	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	10	13.6	30.8	total	5.5	foliage	1.5	0.048701	Harris-Pierce 1994
Selenium	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	21	13.6	29.1	total	5.4	foliage	1.7	0.058419	Harris-Pierce 1994
Selenium	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	30	13.6	24.4	total	5.6	foliage	2	0.081967	Harris-Pierce 1994
Selenium	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	2.5	13.6	28.4	total	5.6	foliage	1.3	0.045775	Harris-Pierce 1994
Selenium	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	5	13.6	33.6	total	5.8	foliage	2.2	0.065476	Harris-Pierce 1994
Selenium	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	10	13.6	30.8	total	5.5	foliage	1.3	0.042208	Harris-Pierce 1994
Selenium	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	21	13.6	29.1	total	5.4	foliage	1.7	0.058419	Harris-Pierce 1994
Selenium	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	30	13.6	24.4	total	5.6	foliage	0.7	0.028689	Harris-Pierce 1994
Selenium	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	2.5	13.6	14.8	total	5.5	foliage	3.9	0.263514	Harris-Pierce 1994
Selenium	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	5	13.6	16	total	5.2	foliage	4	0.25	Harris-Pierce 1994
Selenium	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	10	13.6	15.4	total	5.2	foliage	4.6	0.298701	Harris-Pierce 1994
Selenium	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	21	13.6	15.2	total	5.3	foliage	4.4	0.289474	Harris-Pierce 1994
Selenium	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	30	13.6	16.3	total	5.2	foliage	4.8	0.294479	Harris-Pierce 1994
Selenium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	2.5	13.6	14.8	total	5.5	foliage	3.1	0.209459	Harris-Pierce 1994
Selenium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	13.6	16	total	5.2	foliage	2.9	0.18125	Harris-Pierce 1994
Selenium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	13.6	15.4	total	5.2	foliage	3.2	0.207792	Harris-Pierce 1994
Selenium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	21	13.6	15.2	total	5.3	foliage	3.2	0.210526	Harris-Pierce 1994
Selenium	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	13.6	16.3	total	5.2	foliage	4	0.245399	Harris-Pierce 1994
Selenium	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	2.5	13.6	14.8	total	5.5	foliage	2.7	0.182432	Harris-Pierce 1994
Selenium	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	5	13.6	16	total	5.2	foliage	3	0.1875	Harris-Pierce 1994
Selenium	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	10	13.6	15.4	total	5.2	foliage	3.1	0.201299	Harris-Pierce 1994
Selenium	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	21	13.6	15.2	total	5.3	foliage	3.5	0.230263	Harris-Pierce 1994
Selenium	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	30	13.6	16.3	total	5.2	foliage	3.5	0.214724	Harris-Pierce 1994
Selenium	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	2.5	13.6	14.8	total	5.5	foliage	3.8	0.256757	Harris-Pierce 1994
Selenium	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	5	13.6	16	total	5.2	foliage	3.9	0.24375	Harris-Pierce 1994
Selenium	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	10	13.6	15.4	total	5.2	foliage	4.4	0.285714	Harris-Pierce 1994
Selenium	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	21	13.6	15.2	total	5.3	foliage	4.3	0.282895	Harris-Pierce 1994
Selenium	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	30	13.6	16.3	total	5.2	foliage	4.6	0.282209	Harris-Pierce 1994
Zinc	Arid range	grass	winter wheat	Triticum aestivum	?	multi	variable	?	90.9	total	7.1	foliage	75.8	0.833883	Baxter et al. 1983
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	1210	96.6	total	7.1	foliage	22.1	0.228778	Pierce 1994
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	1210	128	total	7.1	foliage	19.4	0.151563	Pierce 1994
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	15	1210	83.6	total	7	foliage	21.2	0.253589	Pierce 1994
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	20	1210	134	total	6.9	foliage	22.8	0.170149	Pierce 1994

Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	25	1210	116	total	7.1	foliage	16.5	0.142241	Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	1210	126	total	6.9	foliage	22.6	0.179365	Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	35	1210	99.4	total	7.1	foliage	18.2	0.183099	Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	40	1210	165	total	7.1	foliage	21.8	0.132121	Pierce 1994	
Zinc	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	5	1210	96.6	total	7.1	foliage	25.2	0.26087	Pierce 1994	
Zinc	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	10	1210	128	total	7.1	foliage	20.4	0.159375	Pierce 1994	
Zinc	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	15	1210	83.6	total	7	foliage	18.8	0.22488	Pierce 1994	
Zinc	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	20	1210	134	total	6.9	foliage	21.9	0.163433	Pierce 1994	
Zinc	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	25	1210	116	total	7.1	foliage	17.5	0.150862	Pierce 1994	
Zinc	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	30	1210	126	total	6.9	foliage	23.9	0.189683	Pierce 1994	
Zinc	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	35	1210	99.4	total	7.1	foliage	23	0.231388	Pierce 1994	
Zinc	Arid range	grass	bluebunch wheatgras	Agropyron spicatum	2 yr	1	40	1210	165	total	7.1	foliage	26.4	0.16	Pierce 1994	
Zinc	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	5	1210	96.6	total	7.1	foliage	12.6	0.130435	Pierce 1994	
Zinc	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	10	1210	128	total	7.1	foliage	17.4	0.135938	Pierce 1994	
Zinc	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	15	1210	83.6	total	7	foliage	13.8	0.165072	Pierce 1994	
Zinc	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	20	1210	134	total	6.9	foliage	14.5	0.108209	Pierce 1994	
Zinc	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	25	1210	116	total	7.1	foliage	14.3	0.123276	Pierce 1994	
Zinc	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	30	1210	126	total	6.9	foliage	15.4	0.122222	Pierce 1994	
Zinc	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	35	1210	99.4	total	7.1	foliage	18.5	0.186117	Pierce 1994	
Zinc	Arid range	grass	Indian ricegrass	Oryzopsis hymenoide	2 yr	1	40	1210	165	total	7.1	foliage	16	0.09697	Pierce 1994	
Zinc	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	5	1210	96.6	total	7.1	foliage	29.9	0.309524	Pierce 1994	
Zinc	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	10	1210	128	total	7.1	foliage	29.1	0.227344	Pierce 1994	
Zinc	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	15	1210	83.6	total	7	foliage	36.4	0.435407	Pierce 1994	
Zinc	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	20	1210	134	total	6.9	foliage	21.7	0.16194	Pierce 1994	
Zinc	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	25	1210	116	total	7.1	foliage	27.7	0.238793	Pierce 1994	
Zinc	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	30	1210	126	total	6.9	foliage	27.2	0.215873	Pierce 1994	
Zinc	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	35	1210	99.4	total	7.1	foliage	24.9	0.250503	Pierce 1994	
Zinc	Arid range	forb	pulse milkvetch	Astragalus tenellus	2 yr	1	40	1210	165	total	7.1	foliage	27.7	0.167879	Pierce 1994	
Zinc	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	5	1210	96.6	total	7.1	foliage	31.1	0.321946	Pierce 1994	
Zinc	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	10	1210	128	total	7.1	foliage	28.9	0.225781	Pierce 1994	
Zinc	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	15	1210	83.6	total	7	foliage	28.3	0.338517	Pierce 1994	
Zinc	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	20	1210	134	total	6.9	foliage	35.6	0.265672	Pierce 1994	
Zinc	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	25	1210	116	total	7.1	foliage	28.8	0.248276	Pierce 1994	
Zinc	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	30	1210	126	total	6.9	foliage	39.6	0.314286	Pierce 1994	
Zinc	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	35	1210	99.4	total	7.1	foliage	21.3	0.214286	Pierce 1994	
Zinc	Arid range	shrub	big sagebrush	Artemisia tridentata \	2 yr	1	40	1210	165	total	7.1	foliage	37.4	0.226667	Pierce 1994	
Zinc	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	5	1210	96.6	total	7.1	foliage	57.8	0.598344	Pierce 1994	
Zinc	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	10	1210	128	total	7.1	foliage	46.5	0.363281	Pierce 1994	
Zinc	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	15	1210	83.6	total	7	foliage	52.2	0.624402	Pierce 1994	
Zinc	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	20	1210	134	total	6.9	foliage	37.7	0.281343	Pierce 1994	
Zinc	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	25	1210	116	total	7.1	foliage	43.1	0.371552	Pierce 1994	
Zinc	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	30	1210	126	total	6.9	foliage	60.5	0.480159	Pierce 1994	
Zinc	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	35	1210	99.4	total	7.1	foliage	41.4	0.416499	Pierce 1994	
Zinc	Arid range	shrub	Douglas rabbitbrush	Chrysothamnus viscic	2 yr	1	40	1210	165	total	7.1	foliage	41.6	0.252121	Pierce 1994	
Zinc	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	2.5	776	43.8	total	5.6	foliage	23	0.525114	Harris-Pierce 1994	
Zinc	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	5	776	47	total	5.8	foliage	26.8	0.570213	Harris-Pierce 1994	
Zinc	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	10	776	46.5	total	5.5	foliage	20.8	0.447312	Harris-Pierce 1994	
Zinc	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	21	776	43.8	total	5.4	foliage	24.7	0.563927	Harris-Pierce 1994	
Zinc	Arid range	grass	blue grama	Bouteloua gracilis	1 yr	1	30	776	41	total	5.6	foliage	28.1	0.685366	Harris-Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	2.5	776	43.8	total	5.6	foliage	20.6	0.47032	Harris-Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	5	776	47	total	5.8	foliage	22.5	0.478723	Harris-Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	10	776	46.5	total	5.5	foliage	23.6	0.507527	Harris-Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	21	776	43.8	total	5.4	foliage	19.9	0.454338	Harris-Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	1 yr	1	30	776	41	total	5.6	foliage	18.8	0.458537	Harris-Pierce 1994	

Zinc	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	2.5	776	43.8	total	5.6	foliage	15.2	0.347032	Harris-Pierce 1994	
Zinc	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	5	776	47	total	5.8	foliage	17	0.361702	Harris-Pierce 1994	
Zinc	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	10	776	46.5	total	5.5	foliage	17.4	0.374194	Harris-Pierce 1994	
Zinc	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	21	776	43.8	total	5.4	foliage	17.9	0.408676	Harris-Pierce 1994	
Zinc	Arid range	grass	buffalo grass	Buchloe dactyloides	1 yr	1	30	776	41	total	5.6	foliage	17.6	0.429268	Harris-Pierce 1994	
Zinc	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	2.5	776	43.8	total	5.6	foliage	28.8	0.657534	Harris-Pierce 1994	
Zinc	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	5	776	47	total	5.8	foliage	31.6	0.67234	Harris-Pierce 1994	
Zinc	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	10	776	46.5	total	5.5	foliage	26.4	0.567742	Harris-Pierce 1994	
Zinc	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	21	776	43.8	total	5.4	foliage	29.6	0.675799	Harris-Pierce 1994	
Zinc	Arid range	shrub	fringed sage	Artemisia frigida	1 yr	1	30	776	41	total	5.6	foliage	25.7	0.626829	Harris-Pierce 1994	
Zinc	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	2.5	776	19.8	total	5.5	foliage	13.8	0.69697	Harris-Pierce 1994	
Zinc	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	5	776	24.4	total	5.2	foliage	16	0.655738	Harris-Pierce 1994	
Zinc	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	10	776	27.8	total	5.2	foliage	16.6	0.597122	Harris-Pierce 1994	
Zinc	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	21	776	34.8	total	5.3	foliage	17.2	0.494253	Harris-Pierce 1994	
Zinc	Arid range	grass	blue grama	Bouteloua gracilis	2 yr	1	30	776	64.4	total	5.2	foliage	16.9	0.262422	Harris-Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	2.5	776	19.8	total	5.5	foliage	18.5	0.934343	Harris-Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	5	776	24.4	total	5.2	foliage	18.8	0.770492	Harris-Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	10	776	27.8	total	5.2	foliage	18.4	0.661871	Harris-Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	21	776	34.8	total	5.3	foliage	18.3	0.525862	Harris-Pierce 1994	
Zinc	Arid range	grass	western wheatgrass	Agropyron smithii	2 yr	1	30	776	64.4	total	5.2	foliage	21.4	0.332298	Harris-Pierce 1994	
Zinc	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	2.5	776	19.8	total	5.5	foliage	16.9	0.853535	Harris-Pierce 1994	
Zinc	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	5	776	24.4	total	5.2	foliage	19.4	0.795082	Harris-Pierce 1994	
Zinc	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	10	776	27.8	total	5.2	foliage	18.6	0.669065	Harris-Pierce 1994	
Zinc	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	21	776	34.8	total	5.3	foliage	20.5	0.58908	Harris-Pierce 1994	
Zinc	Arid range	grass	buffalo grass	Buchloe dactyloides	2 yr	1	30	776	64.4	total	5.2	foliage	20	0.310559	Harris-Pierce 1994	
Zinc	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	2.5	776	19.8	total	5.5	foliage	19.4	0.979798	Harris-Pierce 1994	
Zinc	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	5	776	24.4	total	5.2	foliage	19.7	0.807377	Harris-Pierce 1994	
Zinc	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	10	776	27.8	total	5.2	foliage	18.8	0.676259	Harris-Pierce 1994	
Zinc	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	21	776	34.8	total	5.3	foliage	18.5	0.531609	Harris-Pierce 1994	
Zinc	Arid range	shrub	fringed sage	Artemisia frigida	2 yr	1	30	776	64.4	total	5.2	foliage	17.6	0.273292	Harris-Pierce 1994	
Zinc	NW forest	tree	hybrid poplar	Populus trichocarpa x	7 yr; 4 pos	1	512	2000	1346.3	nitric-perchl	4.6	foliage	419.1	0.311298	Greenleaf-Jenkins and Zasoski 1986	
Zinc	NW forest	tree	lombardy poplar	Populus nigra	7 yr; 4 pos	1	512	2000	1346.3	nitric-perchl	4.6	foliage	278.1	0.206566	Greenleaf-Jenkins and Zasoski 1987	
Zinc	NW forest	tree	Douglas-fir	Pseudotsuga menzies	7 yr; 4 pos	1	512	2000	1346.3	nitric-perchl	4.6	foliage	52.7	0.039144	Greenleaf-Jenkins and Zasoski 1988	
Zinc	NW forest	tree	Western red cedar	Thuja plicata	7 yr; 4 pos	1	512	2000	1346.3	nitric-perchl	4.6	foliage	100.1	0.074352	Greenleaf-Jenkins and Zasoski 1989	
Zinc	NW forest	tree	Western hemlock	Tsuga heterophylla	7 yr; 4 pos	1	512	2000	1346.3	nitric-perchl	4.6	foliage	33.3	0.024734	Greenleaf-Jenkins and Zasoski 1990	
Zinc	NW forest	tree	Sitka spruce	Picea sitchensis	7 yr; 4 pos	1	512	2000	1346.3	nitric-perchl	4.6	foliage	60.5	0.044938	Greenleaf-Jenkins and Zasoski 1991	
Zinc	NW forest	tree	ponderosa pine	Pinus ponderosa	7 yr; 4 pos	1	512	2000	1346.3	nitric-perchl	4.6	foliage	55.9	0.041521	Greenleaf-Jenkins and Zasoski 1992	
Zinc	NW forest	tree	lombardy poplar	Populus nigra	7 yr; 2-3 p	1	512	2000	1360.7	nitric-perchl	4.7	foliage	443	0.325568	Greenleaf-Jenkins and Zasoski 1993	
Zinc	NW forest	tree	black cottonwood	Populus trichocarpa	7 yr; 2-3 p	1	512	2000	1360.7	nitric-perchl	4.7	foliage	398.8	0.293084	Greenleaf-Jenkins and Zasoski 1994	
Zinc	NW forest	tree	paper birch	Betula paperifera	7 yr; 2-3 p	1	512	2000	1360.7	nitric-perchl	4.7	foliage	1180.4	0.867495	Greenleaf-Jenkins and Zasoski 1995	
Zinc	NW forest	tree	Douglas-fir	Pseudotsuga menzies	7 yr; 2-3 p	1	512	2000	1360.7	nitric-perchl	4.7	foliage	124.5	0.091497	Greenleaf-Jenkins and Zasoski 1996	
Zinc	old field	grass	Japanese brome	Bromus japonicus	10 yr after	50	1.792	865.8-128	107.2	nitric acid	4.4	whole st	80	0.746269	Levine et al. 1989	
Zinc	old field	grass	bluegrass	Poa spp.	10 yr after	50	1.792	865.8-128	107.2	nitric acid	4.4	whole st	85.05	0.793377	Levine et al. 1989	
Zinc	old field	shrub	woody plant	Rubus frondosis	10 yr after	50	1.792	865.8-128	107.2	nitric acid	4.4	1st 10 c	41	0.382463	Levine et al. 1989	
Zinc	old field	grass	foxtail		10 yr after	50	1.792	865.8-128	107.2	nitric acid	4.4	whole st	97.33	0.907929	Levine et al. 1989	
Zinc	old field	grass	blue grass	Poa spp.	about 10 y	50	1.792	19.8	90.25	nitric acid, h	999		85.05	0.942382	Hall et al. 1990	
Zinc	old field	grass	Japanese brome	Bromus japonicus	about 10 y	50	1.792	19.8	90.25	nitric acid, h	999		80	0.886427	Hall et al. 1990	

APPENDIX 2

Published data for use in the derivation of soil-insect contaminant uptake models

Analyte	Study Location	Sample Location	Study type	Species	Soil Conc mg/kg dry wt	Arthropod Conc mg/kg dry wt	Reference
cadmium	Great Britain	control	non-sludge	carnivore arthropods	0.75	2	Hunter and Johnson 1982
cadmium	Great Britain	control	non-sludge	herbivore arthropods	0.75	1.2	Hunter and Johnson 1982
cadmium	Austria	site 5	non-sludge	Blattodea-E. sylvestris	2.5	3.5	Rabitsch 1995a
cadmium	Austria	site 5	non-sludge	chilopoda - E. grossipes	2.5	8	Rabitsch 1995a
cadmium	Austria	site 5	non-sludge	carabidae-C. hortensis	2.5	10.8	Rabitsch 1995a
cadmium	Austria	site 5	non-sludge	staphylinidae-Q. fuliginosus	2.5	6.6	Rabitsch 1995a
cadmium	Austria	site 5	non-sludge	diplopoda-P. complanatus	2.5	2.9	Rabitsch 1995a
cadmium	Austria	site 5	non-sludge	carabidae-P. oblongopunctatus	2.5	2.8	Rabitsch 1995a
cadmium	Austria	site 5	non-sludge	carabidae-P. metallicus	2.5	2.5	Rabitsch 1995a
cadmium	Austria	site 5	non-sludge	isopoda-T. ratzeburgi	2.5	17.7	Rabitsch 1995a
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	millepede-1	2.7	4.5	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	millepede-3	2.7	2.1	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	hornets	2.7	2.3	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	lep. N. c-nigrum	2.7	1.7	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	millepede-2	2.7	3	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	carrion beetles	2.7	0.85	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	lep. C. paleogama	2.7	0.38	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	lep. L. unipuncta	2.7	0.38	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	bark beetle larvae	2.7	1.5	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	lep. Halisidota and spilosoma	2.7	4.6	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	lep. L. dispar	2.7	0.77	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	lep. A. pyramidoides	2.7	0.64	Beyer et al. 1985
cadmium	Pennsylvania	Bake Oven Knob	non-sludge	lep. M. americanum	2.7	0.7	Beyer et al. 1985
cadmium	Great Britain	intermediate	non-sludge	herbivore arthropods	3.1	4.5	Hunter and Johnson 1982
cadmium	Great Britain	intermediate	non-sludge	carnivore arthropods	3.1	6.9	Hunter and Johnson 1982
cadmium	Great Britain	refinery	non-sludge	carnivore arthropods	8.5	11.3	Hunter and Johnson 1982
cadmium	Great Britain	refinery	non-sludge	herbivore arthropods	8.5	10.9	Hunter and Johnson 1982
cadmium	Austria	site 1	non-sludge	staphylinidae-Q. fuliginosus	23.3	19.6	Rabitsch 1995a
cadmium	Austria	site 1	non-sludge	chilopoda - L. tricuspis	23.3	7.9	Rabitsch 1995a
cadmium	Austria	site 1	non-sludge	staphylinidae-P. fossor	23.3	5.7	Rabitsch 1995a
cadmium	Austria	site 1	non-sludge	diplopoda-M. mutabilis	23.3	23.4	Rabitsch 1995a
cadmium	Austria	site 1	non-sludge	isopoda-T. ratzeburgi	23.3	47.7	Rabitsch 1995a
cadmium	Austria	site 1	non-sludge	carabidae-P. versicolor	23.3	4.1	Rabitsch 1995a
cadmium	Austria	site 1	non-sludge	diplopoda-O. pusilla	23.3	34.5	Rabitsch 1995a
cadmium	Austria	site 1	non-sludge	carabidae-P. oblongopunctatus	23.3	12.3	Rabitsch 1995a
cadmium	Pennsylvania	Palmerton	non-sludge	lep. L. dispar	35	3.3	Beyer et al. 1985
cadmium	Austria	site 4	non-sludge	diplopoda-M. mutabilis	35	16.6	Rabitsch 1995a

cadmium	Austria	site 4	non-sludge	staphylinidae-P. fossor	35	16.5	Rabitsch 1995a
cadmium	Austria	site 4	non-sludge	curculionidae-H. abietus	35	1.4	Rabitsch 1995a
cadmium	Austria	site 4	non-sludge	chilopoda - L. forficatus	35	4.8	Rabitsch 1995a
cadmium	Pennsylvania	Palmerton	non-sludge	lep. N. c-nigrum	35	4.7	Beyer et al. 1985
cadmium	Pennsylvania	Palmerton	non-sludge	lep. P. excaecatus	35	8.3	Beyer et al. 1985
cadmium	Pennsylvania	Palmerton	non-sludge	lep. L. unipuncta	35	1.2	Beyer et al. 1985
cadmium	Pennsylvania	Palmerton	non-sludge	lep. C. paleogama	35	0.72	Beyer et al. 1985
cadmium	Pennsylvania	Palmerton	non-sludge	dip. Sarcophagidae	35	44	Beyer et al. 1985
cadmium	Austria	site 4	non-sludge	carabidae-C. hortensis	35	7.6	Rabitsch 1995a
cadmium	Pennsylvania	Palmerton	non-sludge	bark beetle larvae	35	6.2	Beyer et al. 1985
cadmium	Austria	site 4	non-sludge	diplopoda-O. pusilla	35	20.5	Rabitsch 1995a
cadmium	Pennsylvania	Palmerton	non-sludge	lep. A. pyramidoides	35	5.2	Beyer et al. 1985
cadmium	Pennsylvania	Palmerton	non-sludge	centipede	35	28	Beyer et al. 1985
cadmium	Pennsylvania	Palmerton	non-sludge	dip. Calliphoridae	35	29	Beyer et al. 1985
cadmium	Pennsylvania	Palmerton	non-sludge	carrion beetles	35	1.3	Beyer et al. 1985
cadmium	Pennsylvania	Palmerton	non-sludge	lep. H. tessellaris	35	11	Beyer et al. 1985
cadmium	Austria	site 3	non-sludge	isopoda-T. ratzeburgi	72.7	64.9	Rabitsch 1995a
cadmium	Austria	site 3	non-sludge	staphylinidae-Z. humeralis	72.7	7.3	Rabitsch 1995a
cadmium	Austria	site 3	non-sludge	staphylinidae-P. fossor	72.7	9.5	Rabitsch 1995a
cadmium	Austria	site 3	non-sludge	staphylinidae-X. linearis	72.7	8.2	Rabitsch 1995a
cadmium	Austria	site 3	non-sludge	curculionidae-O. raucus	72.7	2.7	Rabitsch 1995a
cadmium	Austria	site 3	non-sludge	diplopoda-O. pusilla	72.7	49.7	Rabitsch 1995a
cadmium	Austria	site 3	non-sludge	staphylinidae-G. circellaris	72.7	16.7	Rabitsch 1995a
cadmium	Austria	site 3	non-sludge	curculionidae-O. ovatus	72.7	8.4	Rabitsch 1995a
cadmium	Austria	sit e 2	non-sludge	curculionidae-O. crategi	82.1	5.5	Rabitsch 1995a
cadmium	Austria	sit e 2	non-sludge	chilopoda - L. forficatus	82.1	4.8	Rabitsch 1995a
cadmium	Austria	sit e 2	non-sludge	curculionidae-O. ovatus	82.1	9.9	Rabitsch 1995a
cadmium	Austria	sit e 2	non-sludge	carabidae-A. lunicollis	82.1	14.9	Rabitsch 1995a
cadmium	Austria	sit e 2	non-sludge	carabidae-H. rufipes	82.1	4.4	Rabitsch 1995a
cadmium	Austria	sit e 2	non-sludge	carabidae-P. versicolor	82.1	18.1	Rabitsch 1995a
cadmium	Austria	sit e 2	non-sludge	carabidae-C. erratus	82.1	7.6	Rabitsch 1995a
cadmium	Austria	sit e 2	non-sludge	isopoda-T. rathkei	82.1	30.2	Rabitsch 1995a
cadmium	Austria	sit e 2	non-sludge	isopoda-P. scaber	82.1	16.3	Rabitsch 1995a
cadmium	Austria	sit e 2	non-sludge	Orthoptera-C. montanus	82.1	12.1	Rabitsch 1995a
cadmium	British Columbia	na	non-sludge	carabidae ad	0.4	0.3	Carter 1983
cadmium	British Columbia	na	non-sludge	millipedes	0.4	0.2	Carter 1983
cadmium	British Columbia	na	non-sludge	staphylinid ad	0.4	0.6	Carter 1983
cd	Poland	Kuzie	non-sludge	herbivore arthropods	0.1	0.8	Dmowski and Karolewski 1979
cd	Poland	Kuzie	non-sludge	predatory arthropds	0.1	2.4	Dmowski and Karolewski 1979
cd	Poland	Kuzie	non-sludge	omnivore arthropods	0.1	1.1	Dmowski and Karolewski 1979

cd	Poland	protection zone	non-sludge	herbivore arthropods	4.1	10.3	Dmowski and Karolewski 1979
cd	Poland	protection zone	non-sludge	omnivore arthropods	4.1	15.3	Dmowski and Karolewski 1979
cd	Poland	protection zone	non-sludge	predatory arthropds	4.1	12.1	Dmowski and Karolewski 1979
cd	Poland	pine forest	non-sludge	herbivore arthropods	7.8	11.3	Dmowski and Karolewski 1979
cd	Poland	pine forest	non-sludge	predatory arthropds	7.8	33.5	Dmowski and Karolewski 1979
cd	Poland	pine forest	non-sludge	omnivore arthropods	7.8	23.9	Dmowski and Karolewski 1979
copper	Great Britain	control	non-sludge	carnivore arthropods	9.3	22.7	Hunter and Johnson 1982
copper	Great Britain	control	non-sludge	herbivore arthropods	9.3	16.9	Hunter and Johnson 1982
copper	Pennsylvania	Palmerton	non-sludge	bark beetle larvae	9.9	22	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	dip. Calliphoridae	9.9	18	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	lep. P. excaecatus	9.9	9.9	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	lep. N. c-nigrum	9.9	15	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	lep. C. paleogama	9.9	15	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	lep. L. unipuncta	9.9	7.2	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	lep. A. pyramidoides	9.9	9.5	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	lep. L. dispar	9.9	13	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	carrion beetles	9.9	13	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	centipede	9.9	41	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	dip. Sarcophagidae	9.9	18	Beyer et al. 1985
copper	Pennsylvania	Palmerton	non-sludge	lep. H. tessellaris	9.9	9.3	Beyer et al. 1985
copper	Austria	sit e 2	non-sludge	isopods	13	187	Wieser et al. 1976
copper	Pennsylvania	Bake Oven Knob	non-sludge	carrion beetles	18	12	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	millepede-1	18	160	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	hornets	18	38	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	lep. L. unipuncta	18	3.8	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	lep. L. dispar	18	13	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	millepede-3	18	150	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	lep. Halisidota and spilosoma	18	6.2	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	millepede-2	18	160	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	lep. M. americanum	18	12	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	lep. C. paleogama	18	13	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	lep. A. pyramidoides	18	5.5	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	lep. N. c-nigrum	18	6.1	Beyer et al. 1985
copper	Pennsylvania	Bake Oven Knob	non-sludge	bark beetle larvae	18	28	Beyer et al. 1985
copper	British Columbia	na	non-sludge	carabidae ad	26	13	Carter 1983
copper	British Columbia	na	non-sludge	millipedes	26	221	Carter 1983
copper	British Columbia	na	non-sludge	staphylinid ad	26	32	Carter 1983
copper	Austria	site 5	non-sludge	Blattodea-E. sylvestris	36.5	16.4	Rabitsch 1995a
copper	Austria	site 5	non-sludge	diplopoda-P. complanatus	36.5	192.6	Rabitsch 1995a
copper	Austria	site 5	non-sludge	chilopoda - E. grossipes	36.5	58.2	Rabitsch 1995a

copper	Austria	site 5	non-sludge	isopoda-T. ratzeburgi	36.5	425.2	Rabitsch 1995a
copper	Austria	site 5	non-sludge	carabidae-P. oblongopunctatus	36.5	20.9	Rabitsch 1995a
copper	Austria	site 5	non-sludge	carabidae-C. hortensis	36.5	14	Rabitsch 1995a
copper	Austria	site 5	non-sludge	carabidae-P. metallicus	36.5	13.66	Rabitsch 1995a
copper	Austria	site 5	non-sludge	staphylinidae-Q. fuliginosus	36.5	12.4	Rabitsch 1995a
copper	Austria	site 4c	non-sludge	isopods	61	184	Wieser et al. 1976
copper	Austria	site 1	non-sludge	isopods	72	115	Wieser et al. 1976
copper	Austria	site 3	non-sludge	isopods	85	74	Wieser et al. 1976
copper	Austria	site 1	non-sludge	carabidae-P. versicolor	113.5	14.9	Rabitsch 1995a
copper	Austria	site 1	non-sludge	isopoda-T. ratzeburgi	113.5	354.6	Rabitsch 1995a
copper	Austria	site 1	non-sludge	chilopoda - L. tricuspis	113.5	31.5	Rabitsch 1995a
copper	Austria	site 1	non-sludge	staphylinidae-P. fossor	113.5	37.3	Rabitsch 1995a
copper	Austria	site 1	non-sludge	staphylinidae-Q. fuliginosus	113.5	14	Rabitsch 1995a
copper	Austria	site 1	non-sludge	diplopoda-M. mutabilis	113.5	183.7	Rabitsch 1995a
copper	Austria	site 1	non-sludge	diplopoda-O. pusilla	113.5	338.5	Rabitsch 1995a
copper	Austria	site 1	non-sludge	carabidae-P. oblongopunctatus	113.5	25.3	Rabitsch 1995a
copper	Austria	site 4	non-sludge	diplopoda-M. mutabilis	129.7	200.4	Rabitsch 1995a
copper	Austria	site 4	non-sludge	carabidae-C. hortensis	129.7	12	Rabitsch 1995a
copper	Austria	site 4	non-sludge	diplopoda-O. pusilla	129.7	413.2	Rabitsch 1995a
copper	Austria	site 4	non-sludge	chilopoda - L. forficatus	129.7	37.7	Rabitsch 1995a
copper	Austria	site 4	non-sludge	curculionidae-H. abietus	129.7	40.2	Rabitsch 1995a
copper	Austria	site 4	non-sludge	staphylinidae-P. fossor	129.7	37.7	Rabitsch 1995a
copper	Great Britain	intermediate	non-sludge	herbivore arthropods	246	78.3	Hunter and Johnson 1982
copper	Great Britain	intermediate	non-sludge	carnivore arthropods	246	86.4	Hunter and Johnson 1982
copper	Austria	site 3	non-sludge	curculionidae-O. raucus	294.9	33.4	Rabitsch 1995a
copper	Austria	site 3	non-sludge	staphylinidae-Z. humeralis	294.9	41.9	Rabitsch 1995a
copper	Austria	site 3	non-sludge	staphylinidae-X. linearis	294.9	14.5	Rabitsch 1995a
copper	Austria	site 3	non-sludge	curculionidae-O. ovatus	294.9	18.4	Rabitsch 1995a
copper	Austria	site 3	non-sludge	staphylinidae-P. fossor	294.9	27.6	Rabitsch 1995a
copper	Austria	site 3	non-sludge	staphylinidae-G. circellaris	294.9	76.4	Rabitsch 1995a
copper	Austria	site 3	non-sludge	isopoda-T. ratzeburgi	294.9	455.4	Rabitsch 1995a
copper	Austria	site 3	non-sludge	diplopoda-O. pusilla	294.9	371.7	Rabitsch 1995a
copper	Austria	site 4a	non-sludge	isopods	540	487	Wieser et al. 1976
copper	Austria	site 5	non-sludge	isopods	570	538	Wieser et al. 1976
copper	Austria	site 4b	non-sludge	isopods	1002	460	Wieser et al. 1976
copper	Austria	site 2	non-sludge	curculionidae-O. crategi	1900.3	9.4	Rabitsch 1995a
copper	Austria	site 2	non-sludge	carabidae-A. lunicollis	1900.3	39.6	Rabitsch 1995a
copper	Austria	site 2	non-sludge	carabidae-H. rufipes	1900.3	15.2	Rabitsch 1995a
copper	Austria	site 2	non-sludge	isopoda-T. rathkei	1900.3	81.7	Rabitsch 1995a
copper	Austria	site 2	non-sludge	Orthoptera-C. montanus	1900.3	136.5	Rabitsch 1995a

copper	Austria	site 2	non-sludge	isopoda-P. scaber	1900.3	185.8	Rabitsch 1995a
copper	Austria	site 2	non-sludge	chilopoda - L. forficatus	1900.3	41.6	Rabitsch 1995a
copper	Austria	site 2	non-sludge	carabidae-P. versicolor	1900.3	18.2	Rabitsch 1995a
copper	Austria	site 2	non-sludge	carabidae-C. erratus	1900.3	28.1	Rabitsch 1995a
copper	Austria	site 2	non-sludge	curculionidae-O. ovatus	1900.3	20.5	Rabitsch 1995a
copper	Great Britain	refinery	non-sludge	herbivore arthropods	2480	310	Hunter and Johnson 1982
copper	Great Britain	refinery	non-sludge	carnivore arthropods	2480	568	Hunter and Johnson 1982
lead	Poland	Kuzie	non-sludge	omnivore arthropods	3	2.3	Dmowski and Karolewski 1979
lead	Poland	Kuzie	non-sludge	herbivore arthropods	3	1.4	Dmowski and Karolewski 1979
lead	Poland	Kuzie	non-sludge	predatory arthropds	3	0	Dmowski and Karolewski 1979
lead	Illinois	lower conc areas	non-sludge	chewing insects	12	3.4	Price et al. 1974
lead	Illinois	lower conc areas	non-sludge	predatory inscets	12	3.3	Price et al. 1974
lead	Illinois	lower conc areas	non-sludge	sucking insects	12	4.7	Price et al. 1974
lead	Illinois	I-74; 13 - 20 m from	non-sludge	sucking insects	20	9.8	Price et al. 1974
lead	Illinois	I-74; 13 - 20 m from	non-sludge	predatory inscets	20	20	Price et al. 1974
lead	Illinois	I-74; 13 - 20 m from	non-sludge	chewing insects	20	10.5	Price et al. 1974
lead	Pennsylvania	Palmerton	non-sludge	bark beetle larvae	41	0.1	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	lep. L. dispar	41	8.7	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	centipede	41	1.2	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	lep. C. paleogama	41	2.1	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	lep. L. unipuncta	41	1.9	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	lep. H. tessellaris	41	6.8	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	lep. A. pyramidoides	41	4.8	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	lep. P. excaecatus	41	3.3	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	lep. N. c-nigrum	41	2.7	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	dip. Calliphoridae	41	15	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	carrion beetles	41	2.5	Beyer et al. 1985
lead	Pennsylvania	Palmerton	non-sludge	dip. Sarcophagidae	41	24	Beyer et al. 1985
lead	Illinois	brownfield meadow 0	non-sludge	chewing insects	45	7.2	Price et al. 1974
lead	Illinois	brownfield meadow 0	non-sludge	predatory inscets	45	22.8	Price et al. 1974
lead	Illinois	brownfield meadow 0	non-sludge	sucking insects	45	4.6	Price et al. 1974
lead	Illinois	I-74; 0.7m from shou	non-sludge	chewing insects	60	27.3	Price et al. 1974
lead	Illinois	I-74; 0.7m from shou	non-sludge	sucking insects	60	15.7	Price et al. 1974
lead	Illinois	I-74; 0.7m from shou	non-sludge	predatory inscets	60	31	Price et al. 1974
lead	Great Britain	control	non-sludge	carabidae	119.6	5.7	Andrews et al. 1989a
lead	Great Britain	control	non-sludge	spiders	119.6	22.2	Andrews et al. 1989a
lead	Great Britain	control	non-sludge	diptera	119.6	28.1	Andrews et al. 1989a
lead	Pennsylvania	Bake Oven Knob	non-sludge	lep. N. c-nigrum	150	1.9	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	millepede-3	150	19	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	lep. M. americanum	150	1.3	Beyer et al. 1985

lead	Pennsylvania	Bake Oven Knob	non-sludge	lep. L. dispar	150	3.4	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	lep. Halisidota and spilosoma	150	2	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	lep. L. unipuncta	150	1	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	bark beetle larvae	150	0.1	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	millepede-2	150	27	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	carrion beetles	150	1.6	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	lep. A. pyramidoides	150	1.7	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	lep. C. paleogama	150	1.5	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	millepede-1	150	22	Beyer et al. 1985
lead	Pennsylvania	Bake Oven Knob	non-sludge	hornets	150	2.1	Beyer et al. 1985
lead	Austria	site 5	non-sludge	carabidae-C. hortensis	574	3.8	Rabitsch 1995a
lead	Austria	site 5	non-sludge	diplopoda-P. complanatus	574	22.9	Rabitsch 1995a
lead	Austria	site 5	non-sludge	carabidae-P. oblongopunctatus	574	12.1	Rabitsch 1995a
lead	Austria	site 5	non-sludge	carabidae-P. metallicus	574	6.1	Rabitsch 1995a
lead	Austria	site 5	non-sludge	chilopoda - E. grossipes	574	12.7	Rabitsch 1995a
lead	Austria	site 5	non-sludge	Blattodea-E. sylvestris	574	34	Rabitsch 1995a
lead	Austria	site 5	non-sludge	isopoda-T. ratzeburgi	574	166.2	Rabitsch 1995a
lead	Austria	site 5	non-sludge	staphylinidae-Q. fuliginosus	574	11.3	Rabitsch 1995a
lead	Poland	protection zone	non-sludge	herbivore arthropods	869.6	108.9	Dmowski and Karolewski 1979
lead	Poland	protection zone	non-sludge	predatory arthropds	869.6	346.3	Dmowski and Karolewski 1979
lead	Poland	protection zone	non-sludge	omnivore arthropods	869.6	362.6	Dmowski and Karolewski 1979
lead	Poland	pine forest	non-sludge	herbivore arthropods	1377.8	497.6	Dmowski and Karolewski 1979
lead	Poland	pine forest	non-sludge	omnivore arthropods	1377.8	487.2	Dmowski and Karolewski 1979
lead	Poland	pine forest	non-sludge	predatory arthropds	1377.8	434.8	Dmowski and Karolewski 1979
lead	Austria	site 1	non-sludge	staphylinidae-P. fossor	1695	20.9	Rabitsch 1995a
lead	Austria	site 1	non-sludge	staphylinidae-Q. fuliginosus	1695	46	Rabitsch 1995a
lead	Austria	site 1	non-sludge	carabidae-P. versicolor	1695	31.2	Rabitsch 1995a
lead	Austria	site 1	non-sludge	chilopoda - L. tricuspis	1695	27.3	Rabitsch 1995a
lead	Austria	site 1	non-sludge	isopoda-T. ratzeburgi	1695	773.2	Rabitsch 1995a
lead	Austria	site 1	non-sludge	diplopoda-O. pusilla	1695	124.2	Rabitsch 1995a
lead	Austria	site 1	non-sludge	carabidae-P. oblongopunctatus	1695	43.2	Rabitsch 1995a
lead	Austria	site 1	non-sludge	diplopoda-M. mutabilis	1695	124.7	Rabitsch 1995a
lead	Great Britain	tailings dam	non-sludge	diptera	3960	549	Andrews et al. 1989a
lead	Great Britain	tailings dam	non-sludge	spiders	3960	462	Andrews et al. 1989a
lead	Great Britain	tailings dam	non-sludge	carabidae	3960	106	Andrews et al. 1989a
lead	Austria	site 4	non-sludge	staphylinidae-P. fossor	4665	57.8	Rabitsch 1995a
lead	Austria	site 4	non-sludge	curculionidae-H. abietus	4665	28.2	Rabitsch 1995a
lead	Austria	site 4	non-sludge	carabidae-C. hortensis	4665	8.5	Rabitsch 1995a
lead	Austria	site 4	non-sludge	diplopoda-M. mutabilis	4665	271.3	Rabitsch 1995a
lead	Austria	site 4	non-sludge	chilopoda - L. forficatus	4665	60	Rabitsch 1995a

lead	Austria	site 4	non-sludge	diplopoda-O. pusilla	4665	139.5	Rabitsch 1995a
lead	Austria	site 3	non-sludge	staphylinidae-P. fossor	8928	51.8	Rabitsch 1995a
lead	Austria	site 3	non-sludge	curculionidae-O. ovatus	8928	353	Rabitsch 1995a
lead	Austria	site 3	non-sludge	staphylinidae-X. linearis	8928	76.6	Rabitsch 1995a
lead	Austria	site 3	non-sludge	diplopoda-O. pusilla	8928	390.8	Rabitsch 1995a
lead	Austria	site 3	non-sludge	staphylinidae-Z. humeralis	8928	107	Rabitsch 1995a
lead	Austria	site 3	non-sludge	isopoda-T. ratzeburgi	8928	1145.7	Rabitsch 1995a
lead	Austria	site 3	non-sludge	staphylinidae-G. circellaris	8928	1037	Rabitsch 1995a
lead	Austria	site 3	non-sludge	curculionidae-O. raucus	8928	94.5	Rabitsch 1995a
lead	Austria	site 2	non-sludge	carabidae-A. lunicollis	93768	623	Rabitsch 1995a
lead	Austria	site 2	non-sludge	carabidae-P. versicolor	93768	472	Rabitsch 1995a
lead	Austria	site 2	non-sludge	curculionidae-O. ovatus	93768	726	Rabitsch 1995a
lead	Austria	site 2	non-sludge	Orthoptera-C. montanus	93768	306.2	Rabitsch 1995a
lead	Austria	site 2	non-sludge	carabidae-H. rufipes	93768	215	Rabitsch 1995a
lead	Austria	site 2	non-sludge	chilopoda - L. forficatus	93768	29.4	Rabitsch 1995a
lead	Austria	site 2	non-sludge	curculionidae-O. crategi	93768	292	Rabitsch 1995a
lead	Austria	site 2	non-sludge	isopoda-P. scaber	93768	123.8	Rabitsch 1995a
lead	Austria	site 2	non-sludge	carabidae-C. erratus	93768	617	Rabitsch 1995a
lead	Austria	site 2	non-sludge	isopoda-T. rathkei	93768	386.1	Rabitsch 1995a
zinc	Great Britain	control	non-sludge	diptera	61.9	198	Andrews et al. 1989b
zinc	Great Britain	control	non-sludge	spiders	61.9	529	Andrews et al. 1989b
zinc	Great Britain	control	non-sludge	carabidae	61.9	120	Andrews et al. 1989b
zinc	British Columbia	na	non-sludge	staphylinid ad	83	278	Carter 1983
zinc	British Columbia	na	non-sludge	millipedes	83	321	Carter 1983
zinc	British Columbia	na	non-sludge	carabidae ad	83	116	Carter 1983
zinc	Pennsylvania	Bake Oven Knob	non-sludge	lep. L. dispar	230	170	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	lep. A. pyramidoides	230	140	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	lep. N. c-nigrum	230	300	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	lep. L. unipuncta	230	190	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	lep. Halisidota and spilosoma	230	340	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	carrion beetles	230	120	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	bark beetle larvae	230	470	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	millepede-2	230	370	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	lep. M. americanum	230	150	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	millepede-1	230	980	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	lep. C. paleogama	230	160	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	hornets	230	180	Beyer et al. 1985
zinc	Pennsylvania	Bake Oven Knob	non-sludge	millepede-3	230	320	Beyer et al. 1985
zinc	Austria	site 5	non-sludge	carabidae-P. oblongopunctatus	329	125	Rabitsch 1995a
zinc	Austria	site 5	non-sludge	carabidae-C. hortensis	329	104	Rabitsch 1995a

zinc	Austria	site 5	non-sludge	carabidae-P. metallicus	329	157	Rabitsch 1995a
zinc	Austria	site 5	non-sludge	chilopoda - E. grossipes	329	519	Rabitsch 1995a
zinc	Austria	site 5	non-sludge	Blattodea-E. sylvestris	329	267	Rabitsch 1995a
zinc	Austria	site 5	non-sludge	isopoda-T. ratzeburgi	329	857.5	Rabitsch 1995a
zinc	Austria	site 5	non-sludge	staphylinidae-Q. fuliginosus	329	314	Rabitsch 1995a
zinc	Austria	site 5	non-sludge	diplopoda-P. complanatus	329	599	Rabitsch 1995a
zinc	Austria	site 1	non-sludge	carabidae-P. versicolor	1697	247	Rabitsch 1995a
zinc	Austria	site 1	non-sludge	carabidae-P. oblongopunctatus	1697	298	Rabitsch 1995a
zinc	Austria	site 1	non-sludge	staphylinidae-Q. fuliginosus	1697	380	Rabitsch 1995a
zinc	Austria	site 1	non-sludge	isopoda-T. ratzeburgi	1697	1168	Rabitsch 1995a
zinc	Austria	site 1	non-sludge	diplopoda-M. mutabilis	1697	2044	Rabitsch 1995a
zinc	Austria	site 1	non-sludge	chilopoda - L. tricuspis	1697	773	Rabitsch 1995a
zinc	Austria	site 1	non-sludge	diplopoda-O. pusilla	1697	1687	Rabitsch 1995a
zinc	Austria	site 1	non-sludge	staphylinidae-P. fossor	1697	635	Rabitsch 1995a
zinc	Great Britain	tailings dam	non-sludge	spiders	1925	657	Andrews et al. 1989b
zinc	Great Britain	tailings dam	non-sludge	carabidae	1925	340	Andrews et al. 1989b
zinc	Great Britain	tailings dam	non-sludge	diptera	1925	852	Andrews et al. 1989b
zinc	Pennsylvania	Palmerton	non-sludge	centipede	2900	1180	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	carrion beetles	2900	150	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	lep. N. c-nigrum	2900	480	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	lep. P. excaecatus	2900	380	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	lep. L. unipuncta	2900	320	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	lep. H. tessellaris	2900	500	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	lep. C. paleogama	2900	250	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	lep. A. pyramidoides	2900	540	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	lep. L. dispar	2900	280	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	dip. Calliphoridae	2900	740	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	bark beetle larvae	2900	1450	Beyer et al. 1985
zinc	Pennsylvania	Palmerton	non-sludge	dip. Sarcophagidae	2900	460	Beyer et al. 1985
zinc	Austria	site 4	non-sludge	diplopoda-M. mutabilis	5813	1749	Rabitsch 1995a
zinc	Austria	site 4	non-sludge	diplopoda-O. pusilla	5813	984	Rabitsch 1995a
zinc	Austria	site 4	non-sludge	chilopoda - L. forficatus	5813	606	Rabitsch 1995a
zinc	Austria	site 4	non-sludge	carabidae-C. hortensis	5813	113	Rabitsch 1995a
zinc	Austria	site 4	non-sludge	staphylinidae-P. fossor	5813	589	Rabitsch 1995a
zinc	Austria	site 4	non-sludge	curculionidae-H. abietus	5813	78.8	Rabitsch 1995a
zinc	Austria	site 3	non-sludge	diplopoda-O. pusilla	7795	1599	Rabitsch 1995a
zinc	Austria	site 3	non-sludge	staphylinidae-Z. humeralis	7795	95.5	Rabitsch 1995a
zinc	Austria	site 3	non-sludge	staphylinidae-X. linearis	7795	550	Rabitsch 1995a
zinc	Austria	site 3	non-sludge	curculionidae-O. ovatus	7795	98.7	Rabitsch 1995a
zinc	Austria	site 3	non-sludge	staphylinidae-P. fossor	7795	472	Rabitsch 1995a

zinc	Austria	site 3	non-sludge	curculionidae-O. raucus	7795	80.3	Rabitsch 1995a
zinc	Austria	site 3	non-sludge	staphylinidae-G. circellaris	7795	891	Rabitsch 1995a
zinc	Austria	site 3	non-sludge	isopoda-T. ratzeburgi	7795	2430	Rabitsch 1995a
zinc	Austria	site 2	non-sludge	Orthoptera-C. montanus	8050	701	Rabitsch 1995a
zinc	Austria	site 2	non-sludge	isopoda-T. rathkei	8050	150	Rabitsch 1995a
zinc	Austria	site 2	non-sludge	isopoda-P. scaber	8050	970	Rabitsch 1995a
zinc	Austria	site 2	non-sludge	chilopoda - L. forficatus	8050	1064	Rabitsch 1995a
zinc	Austria	site 2	non-sludge	carabidae-P. versicolor	8050	394	Rabitsch 1995a
zinc	Austria	site 2	non-sludge	carabidae-A. lunicollis	8050	402	Rabitsch 1995a
zinc	Austria	site 2	non-sludge	carabidae-H. rufipes	8050	287	Rabitsch 1995a
zinc	Austria	site 2	non-sludge	curculionidae-O. crategi	8050	205.6	Rabitsch 1995a
zinc	Austria	site 2	non-sludge	curculionidae-O. ovatus	8050	137.9	Rabitsch 1995a
zinc	Austria	site 2	non-sludge	carabidae-C. erratus	8050	475	Rabitsch 1995a
zinc	Poland	protection zone	non-sludge	predatory arthropods	80.7	789.6	Dmowski and Karolewski 1979
zinc	Poland	protection zone	non-sludge	herbivore arthropods	80.7	661.5	Dmowski and Karolewski 1979
zinc	Poland	protection zone	non-sludge	omnivore arthropods	80.7	496.5	Dmowski and Karolewski 1979
zinc	Poland	pine forest	non-sludge	predatory arthropods	635.6	1537.8	Dmowski and Karolewski 1979
zinc	Poland	pine forest	non-sludge	herbivore arthropods	635.6	728.4	Dmowski and Karolewski 1979
zinc	Poland	pine forest	non-sludge	omnivore arthropods	635.6	526.1	Dmowski and Karolewski 1979
cadmium	old field, OH sludge		sludge	cabbage looper moth	2.43	1.6	Larsen et al. 1994
cadmium	old field, OH sludge		sludge	blackfaced leafhopper	2.43	5.3	Larsen et al. 1994
cadmium	old field, OH control		sludge	cabbage looper moth	0.26	0.6	Larsen et al. 1994
cadmium	old field, OH control		sludge	blackfaced leafhopper	0.26	1.8	Larsen et al. 1994
cadmium	old field, OH control		sludge	spiders	0.5	2.2	Benninger-Truax and Taylor 1993
cadmium	old field, OH sludge		sludge	spiders	2.2	4.2	Benninger-Truax and Taylor 1993
cadmium	old field, OH control		sludge	crickets	0.5	0.8	Benninger-Truax and Taylor 1993
cadmium	old field, OH sludge		sludge	crickets	2.2	11.4	Benninger-Truax and Taylor 1993
copper	old field, OH sludge		sludge	cabbage looper moth	30.1	19	Larsen et al. 1994
copper	old field, OH sludge		sludge	blackfaced leafhopper	30.1	18	Larsen et al. 1994
copper	old field, OH control		sludge	cabbage looper moth	9.71	18	Larsen et al. 1994
copper	old field, OH control		sludge	blackfaced leafhopper	9.71	16	Larsen et al. 1994
copper	old field, OH control		sludge	spiders	9.6	78.9	Benninger-Truax and Taylor 1993
copper	old field, OH sludge		sludge	spiders	26.7	102.7	Benninger-Truax and Taylor 1993
copper	old field, OH control		sludge	crickets	9.6	33.9	Benninger-Truax and Taylor 1993
copper	old field, OH sludge		sludge	crickets	26.7	47	Benninger-Truax and Taylor 1993
lead	old field, OH sludge		sludge	cabbage looper moth	0.74	4.5	Larsen et al. 1994
lead	old field, OH sludge		sludge	blackfaced leafhopper	0.74	4.8	Larsen et al. 1994
lead	old field, OH control		sludge	cabbage looper moth	0.36	1.75	Larsen et al. 1994
lead	old field, OH control		sludge	blackfaced leafhopper	0.36	2.2	Larsen et al. 1994
lead	old field, OH control		sludge	spiders	23	3.8	Benninger-Truax and Taylor 1993

lead	old field, OH sludge		sludge	spiders	41.2	6.7	Benninger-Truax and Taylor 1993
lead	old field, OH control		sludge	crickets	23	5.8	Benninger-Truax and Taylor 1993
lead	old field, OH sludge		sludge	crickets	41.2	4.7	Benninger-Truax and Taylor 1993
zinc	old field, OH sludge		sludge	cabbage looper moth	9.54	187	Larsen et al. 1994
zinc	old field, OH sludge		sludge	blackfaced leafhopper	9.54	265	Larsen et al. 1994
zinc	old field, OH control		sludge	cabbage looper moth	5.7	175	Larsen et al. 1994
zinc	old field, OH control		sludge	blackfaced leafhopper	5.7	240	Larsen et al. 1994
zinc	old field, OH control		sludge	spiders	62.4	298.3	Benninger-Truax and Taylor 1993
zinc	old field, OH sludge		sludge	spiders	109.2	452.3	Benninger-Truax and Taylor 1993
zinc	old field, OH control		sludge	crickets	62.4	594.6	Benninger-Truax and Taylor 1993
zinc	old field, OH sludge		sludge	crickets	109.2	658.6	Benninger-Truax and Taylor 1993
PCBs	lab exposure		non-sludge	crickets	100	11.11	Paine et al. 1993
PCBs	lab exposure		non-sludge	crickets	250	47.65	Paine et al. 1993
PCBs	lab exposure		non-sludge	crickets	500	92.12	Paine et al. 1993
PCBs	lab exposure		non-sludge	crickets	1000	148.6	Paine et al. 1993
PCBs	lab exposure		non-sludge	crickets	2000	143.9	Paine et al. 1993